Denitrification Kinetics in Anoxic/Aerobic Activated Sludge Systems

by

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Master's Thesis

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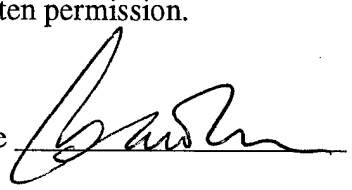
Signature 

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GLOSSARY

ADVSS	Active Denitrifying Volatile Suspended Solids
A/O	Anoxic-Aerobic
ASM	Activated Sludge Model
AVSS	Active Volatile Suspended Solids
BOD	Biological Oxygen Demand
BNR	Biological Nitrogen Removal
BPR	Biological Phosphorous Removal
BSCOD	Biodegradable Soluble COD
CF	Cubic Feet
COD	Chemical Oxygen Demand
DO	Dissolved Oxygen
DPAO	Denitrifying Phosphorus Accumulating Organism
EATS	Enhanced Biological Nutrient Removal
EPA	Environmental Protection Agency
FCOD	Flocculated COD
F/M	Food to mass ratio
HRT	Hydraulic Residence Time
IAWQ	International Association of Water Quality
MCRT	Mean Cell Residence Time
MLE	Modified Ludzack-Ettinger Wastewater Treatment Process
MLSS	Mixed Liquor Suspended Solids
MLVSS	Mixed Liquor Volatile Suspended Solids
N	Nitrogen
ND	Nitrification - Denitrification
NDBEPR	Nitrification-Denitrification Biological Excess Phosphorous Removal
NRT	Nominal Residence Time
NUR	Nitrate Utilization Rate
ORP	Oxidation-Reduction Potential
OUR	Oxygen Utilization Rate
PAO	Phosphorus Accumulating Organism
PHB	Polyhydroxylbutyrate
PN/CD	Partial Nitrification/Complete Denitrification
PP	Polyphosphates
RAS	Return Activated Sludge
RBCOD	Readily Biodegradable Chemical Oxygen Demand
SBCOD	Slowly Biodegradable Chemical Oxygen Demand
SCOD	Soluble COD
SEOUR	Specific Endogenous Oxygen Utilization Rate
SNDR	Specific Denitrification Rate
SRT	Sludge Residence Time
SUR	Specific Utilization Rate

TBSCOD	Total Biodegradable Soluble COD
TN	Total Nitrogen
TSS	Total Suspended Solids
UCBOD	Unbiodegradable COD
VFA	Volatile Fatty Acids
VSS	Volatile Suspended Solids
WWTP	Wastewater Treatment Plant

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Chapter 1 Introduction and Objectives

Nitrogen removal needs at municipal wastewater treatment plants (WWTPs) have increased due to greater concerns about eutrophication and increased interest in reuse of treated municipal effluents. Biological processes are the most cost-effective method for nitrogen removal. Biological nitrogen removal is accomplished in two distinctly different processes by the conversion of nitrogen in the wastewater from organic nitrogen and ammonia to nitrate, followed by reduction of the nitrate to nitrogen gas. Nitrate production occurs in an aerobic activated sludge treatment zone during a process called nitrification. The nitrate is then converted through a series of intermediate steps to nitrogen gas in an anoxic zone (an anaerobic condition with nitrate present) during a process called denitrification, effectively removing the nitrogen from the wastewater. Many different WWTP designs have been developed to incorporate these two conditions for nitrogen removal.

Anoxic-Aerobic (A/O) activated sludge wastewater treatment systems are extensively used at municipal wastewater treatment plants (WWTP) for nitrogen removal. This system consists of an anoxic tank followed by an aerobic tank in the activated sludge process. The influent wastewater and return activated sludge from the secondary clarifier are combined and mixed in the anoxic tank at detention times ranging from 1 to 4 hours. The mixed liquor is aerated and mixed in the aerobic zone with detention times that may range from 4 to 20 hours, depending on the plant design, wastewater characteristics, and operating temperature. The nitrate produced in the aerobic zone is fed to the anoxic zone by a recycle stream at a flow rate that may be 2 to 4 times the influent flow rate where it is converted to nitrogen gas. Bacteria that can use nitrate as an electron acceptor in the anoxic zone oxidize the organic material within the wastewater. Key design parameters that determine the amount of nitrogen removal are the anoxic and aerobic zone detention times, the mixed liquor concentration, and the internal recycle rate. Anoxic zones have been designed as single stage, completely mixed tanks or as a number of completely

mixed tanks in series with different detention times. These designs have been based on empirical equations or a mechanistic activated sludge model.

A common design approach used, dating from early A/O system designs, is to base the anoxic zone design on a specific denitrification rate (SDNR) obtained from an empirical equation relating SDNR to the food to mass (F/M) loading ratio of the anoxic zone. The F/M ratio is equal to the BOD mass loading per day per mass of mixed liquor volatile suspended solids (MLVSS) in the anoxic zone. This equation was based on full-scale plant and pilot plant results but ignores site specific wastewater parameters such as non-biodegradable volatile solids and the wastewater distribution of the influent organic material which has been more recently described in terms of particulate degradable COD, slowly degradable COD, and readily biodegradable soluble COD

More recently, mechanistic models have been developed that relate denitrification kinetic fundamentals to wastewater and plant design parameters. The most popular model is the International Association of Water Quality (IAWQ) model developed by a worldwide committee of activated sludge treatement experts. A form of this model that is generally accepted is contained in a commercially available model form called the BioWin® Model. This model accounts for carbonaceous removal, nitrification, denitrification, and biological phosphorus removal in activated sludge systems. The model predicts the population distribution and substrate removal kinetics of bacteria responsible for biological phosphorus removal, nitrification, and denitrification. The model contains a large number of coefficients, including 23 kinetic and 32 stoichiometric coefficients, and many of these are related to denitrification kinetics and the performance of anoxic zone reactors. Default coefficient values are recommended for the model with site calibration recommended. A key parameter in the model is the influent readily biodegradable COD, which greatly affects anoxic zone COD oxidation rates and thus the denitrification rates.

There is little information that relates denitrification rates and substrate utilization kinetics in anoxic zones to wastewater characteristics at specific WWTP locations. The

variability of denitrification kinetic parameters for different WWTP sites has not been compared so that the general applicability of the values in the IAWQ model is uncertain, as well as the ability to use SDNRs based on the empirical F/M equation. Additionally, little work has been done to investigate denitrification rates in shorter detention time anoxic zones that would be found in staged anoxic reactors.

This research explores the factors affecting anoxic zone specific denitrification rates and determines how SDNRs or denitrification kinetics varied for different municipal wastewaters. These factors include the wastewater characteristics, the solids retention time (SRT) of the activated sludge mixed liquor, the anoxic zone contact time, and activated sludge plant design. An anoxic experimental reactor was operated at four different municipal WWTPs in the Seattle area to meet the research goals. SNDRs were determined under different loading conditions using the wastewater and activated sludge from each of the WWTPs. The SDNRs were adjusted to reflect the estimated active denitrifying biomass driving the denitrification rates. The experimental data from these site tests showed the SDNRs observed at each of the four WWTPs followed a Michaelis-Menton relationship. The experimental data from all four site tests also demonstrated a Monod growth relationship very similar to the relationship predicted by the IAWQ model using the model's default values.

Chapter 2: Literature Review

2.1 Introduction

Biological nitrogen removal is commonly incorporated in municipal wastewater activated sludge treatment processes. A common activated sludge process used for biological nitrogen removal consists of an anoxic tank followed by an aerobic tank. The anoxic tanks provide conditions favorable for bacteria to use nitrate instead of oxygen as a terminal electron acceptor during organic substrate consumption. The aerobic tank provides oxygenated conditions that allow the nitrifying bacteria to convert the organic nitrogen and ammonia within the influent wastewater to nitrate. A portion of the nitrified mixed liquor is recycled back to the anoxic tank, providing nitrate for the denitrification step. This single sludge system will be referred to as an anoxic-aerobic system (A/O) activated sludge system. Design methodology for the system varies from the use of empirical rate equations to detailed mechanistic modeling.

This chapter includes a literature review on fundamental mechanisms of biological nitrification and denitrification, a description of biological nitrogen removal processes, a summary of an empirical design approach used to design denitrification reactors in the activated sludge process, and a review of comprehensive mechanistic modeling approach that can be used to design A/O systems.

2.2 Nitrogen Cycle

The nitrogen cycle describes the aerobic and anaerobic nitrogen transformations that occur in nature and the same processes can be found in biological wastewater treatment. Thus, a description of the nitrogen cycle provides a useful background for understanding nitrogen transformations in wastewater treatment. The biological nitrogen cycle consists of the following biotransformations of nitrogenous compounds: deamination (mineralization), assimilation, denitrification, nitrogen fixation, and nitrification.

Deamination is the biological conversion of nitrogen in organic compounds into the mineral or inorganic form of nitrogen. Assimilation is the process of bacteria incorporating nitrogen into new cell growth. Denitrification is the biological reduction of nitrate to nitrite, nitrogen gas or ammonia. It can be described as assimilative or dissimilative nitrate reduction. Assimilation or synthesis is the biochemical mechanism that uses ammonium or nitrate compounds to form more cell material. Another source of ammonia nitrogen for cell synthesis is from nitrogen gas using a process termed nitrogen fixation. Fixation is predominantly mediated by specialized organism such as certain blue-green algae and only occurs when insufficient quantities of ammonia nitrogen are present for cell synthesis. A limited number of bacteria are able to covert nitrogen gas to ammonia. Nitrification is the biological oxidation of ammonia to nitrite and nitrate. Nitrification occurs in two sequential steps with oxidation of ammonia to nitrite, followed by the oxidation of nitrite to nitrate. Nitrification is carried out by autotrophic bacteria, which use inorganic carbon (CO_2) as their carbon source during cell synthesis. Nitrate is generally the preferred form of nitrogen for plant growth. Denitrification is the biological process where nitrate or nitrite (NO_x) is used for substrate oxidation during cell growth and substrate metabolism in the absence of oxygen. The NO_x is reduced to N_2O or N_2 gas.

The nitrogen cycle is normally in balance as whole, with local imbalances due to auto exhaust, industrial processes, agricultural land practices, and municipal wastewater effluent. The atmosphere is considered the sink for nitrogen. The extent of local nitrogen imbalances caused by municipal wastewater effluent is a function of both the influent wastewater characteristics and the WWTP processes and operating conditions used.

Municipal wastewaters generally have a high nitrogen concentration compared to the streams, lakes, and other bodies receiving wastewater treatment plant effluent. The nitrogen found in municipal wastewater is principally in the form of ammonia or organic

nitrogen, and is in both soluble and particulate form. The soluble form is from urea and amino acids. Untreated wastewater usually contains little or no nitrite or nitrate. Actual nitrogen constituents and their concentrations within municipal wastewaters vary greatly with location both daily and seasonally for a given wastewater. Typical municipal wastewater characteristics are provided in Table 2.1.

Table 2.1 Typical Municipal Wastewater Characteristics (Metcalf and Eddy, 1991)

		Concentration		
Contaminants	Units	Weak	Medium	Strong
Total Solids	mg/L	350	720	1200
Biological Oxygen Demand (BOD)	mg/L	110	220	400
Total Organic Carbon (TOC)	mg/L	80	160	290
Chemical Oxygen Demand (COD)	mg/L	250	500	1000
Total Nitrogen	mg/L	20	40	85
Organic Nitrogen	mg/L	8	15	35
Free Ammonia Nitrogen	mg/L	12	25	50
Nitrites	mg/L	0	0	0
Nitrates	mg/L	0	0	0
Phosphorus	mg/L	4	8	15
Alkalinity	mg/L	50	100	150

Figure 2.1 below shows the nitrogen transformations possible within biological treatment processes (Metcalf & Eddy, 1991).

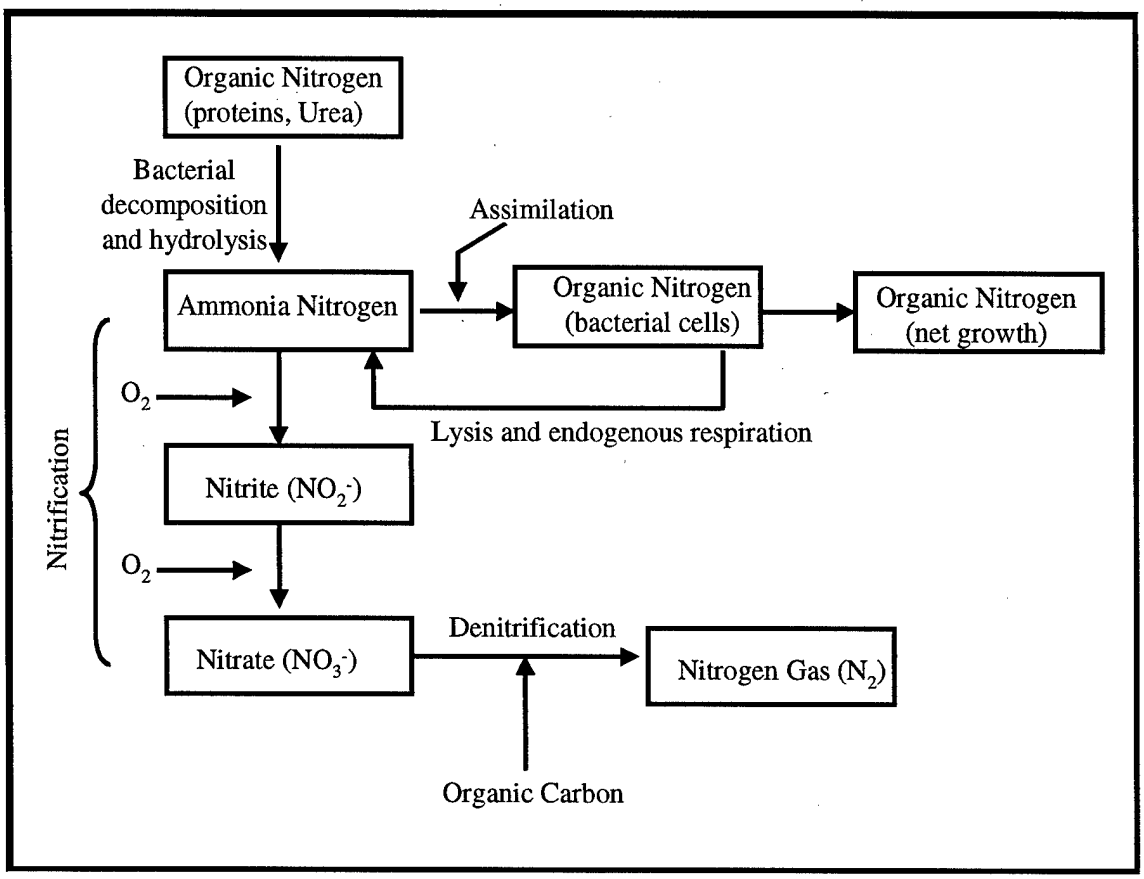

Figure 2.1 Nitrogen Transformations in Biological Treatment Processes

Understanding the nitrogen cycle within WWTPs is vital to understanding the nitrogen removal process. The two principal mechanisms for nitrogen removal are assimilation and the nitrification-denitrification process within biological treatment processes (Figure 2.1). Nitrogen is used as a nutrient during cell growth by microbes, assimilating ammonia-nitrogen and incorporating it into the biomass produced. A portion of ammonia-nitrogen incorporated into new cell growth is released as a result of cell death and lysis. This material is collectively called endogenous residue. The nitrification-denitrification process is a sequential two-step process. In the first step ammonia is converted to nitrate through nitrification. In the second step, nitrate is converted to nitrogen gas during denitrification releasing the nitrogen to the atmosphere. The fundamentals of these two biological reactions are discussed in more detail in the later sections.

2.3 Biological Nitrogen Removal (BNR)

BNR is a two step wastewater treatment process of nitrification and denitrification. Two typically used BNR processes are the Modified Ludzack-Ettinger (MLE) and Bardenpho Designs (Metcalf & Eddy, 1991).

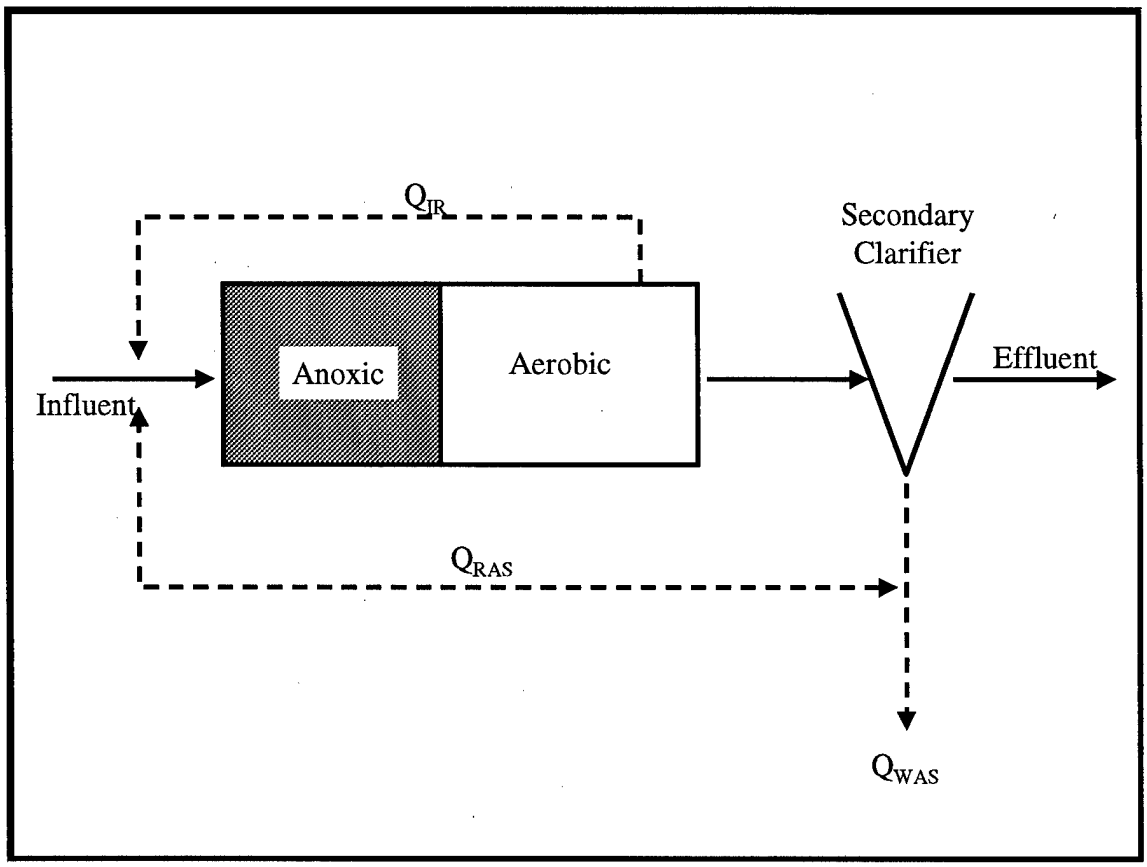

Figure 2.2 Schematic of the Modified Ludzack-Ettinger Wastewater Treatment Process

The MLE design shown in Figure 2.2 is a simple nitrification-denitrification treatment process. This design uses separate reaction zones for carbon oxidation and nitrification and denitrification. Anoxic denotes an environment in which either nitrate or nitrite is present and the dissolved oxygen (DO) concentration is low or zero (Burdick et al., 1982). The wastewater initially enters an anoxic zone to which nitrified mixed liquor is

recycled from the subsequent aerobic nitrification zone. The carbon present in the wastewater is used to denitrify the recycled nitrate. The ammonia in the wastewater passes through the anoxic zone and is nitrified in the aeration basin. The MLE design was one of the first designs to specifically address nitrogen removal. The Bardenpho Process is a more complex design that was intended to increase nitrogen removal performance beyond that of the MLE design.

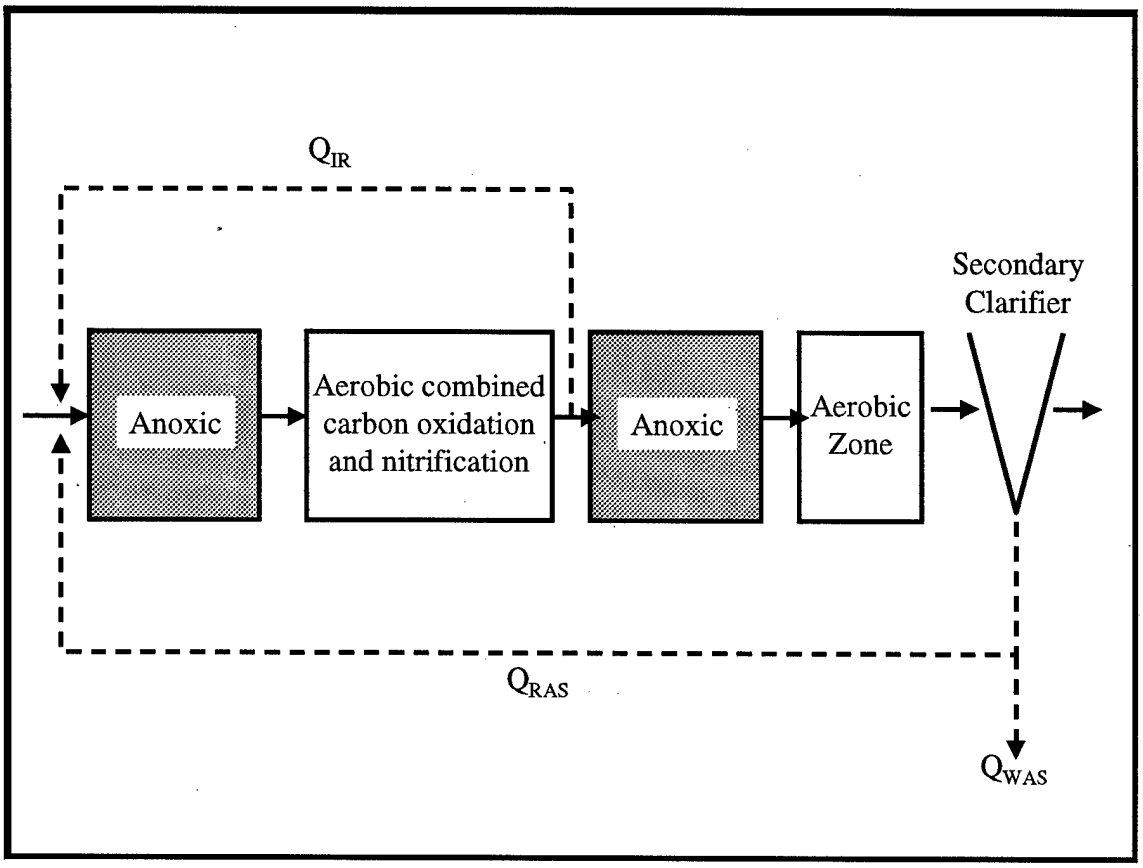

Figure 2.3 Schematic of a Four-Stage Bardenpho Wastewater Treatment Process

The Bardenpho process shown in Figure 2.3 uses both the carbon in the untreated wastewater and carbon from endogenous decay to achieve denitrification (Metcalf & Eddy, 1991). Like the MLE process, this design uses separate reaction zones for carbon oxidation and nitrification and denitrification. The first two stages of the Bardenpho process are identical the MLE process. The nitrified mixed liquor from the first aeration

basin passes into a second anoxic zone. Additional denitrification occurs in the second anoxic zone using endogenous respiration to drive the nitrate reduction rates. These observed denitrification rates in the second anoxic zone are much lower than the rates achieved in the first anoxic zone. The last aerobic zone has a relatively short aeration time of (20-30 minutes) and is used to remove entrained nitrogen gas and nitrify remaining ammonia.

The following two sections describe the fundamental principals of the nitrification and denitrification processes that are critical to the MLE, Bardenpho, and other BNR processes.

2.4 Nitrification Fundamentals

In the BNR nitrification step autotrophic bacteria are assumed to oxidize ammonia sequentially to nitrite and then to nitrate. Research by Focht and Chang (1975) indicated heterotrophic nitrification is possible with bacteria, fungi, and actinomycete genera. However, Randall et al. (1992) stated it is doubtful that significant quantities of nitrate are generated by heterotrophic organisms because their reported maximum specific growth rates are one-tenth that of autotrophic bacteria genera. However, they also noted that under atypical condition such as very alkaline or acidic pH conditions heterotrophic nitrification may be more prominent. Many different autotrophic bacteria have been found which can nitrify such as: *Nitrosomonas*, *Nitrobacter*, *Nitrosococcus*, *Nitrospirra*, *Nitrosocyctis*, and *Nitrosoglea*. Nitrification within wastewater treatment processes has been attributed primarily to *Nitrosomonas* and *Nitrobacter*. However, recent research by Wagner et al. (1996) indicated other types of bacteria may actually dominate in municipal WWTP nitrification processes. They developed an *in situ* identification technique for ammonia and nitrite-oxidizing bacteria that provided a more accurate microbiological identification method based on16SRNA analysis. Their method was used to search for the abundance of *Nitrosomonas* and *Nitrobacter* at nine different WWTPs of which eight had observed nitrification. *Nitrobacter* were not identified in the samples taken at these

nine plants, indicating that if these bacteria were present, they were at concentrations below the detection limit (10^4/mL). Wagner et al (1996) indicated that different DNA probes that can be used with their *in situ* method to identify the presence of other types of autotrophic bacteria in municipal WWTPs are under development. The authors indicated that other genera such as *Nitrococcus, Nitrospina, and Nitrospira* might have been present at high concentrations. Future research in this area may prove which bacteria other than *Nitrosomonas* and *Nitrobacter* are responsible for nitrifying the wastewater within WWTPs.

The generally accepted energy yielding two-step-oxidation of ammonia to nitrate is as follows (Randall et al., 1992):

Nitrosomonas

$$2NH_4^+ + 3O_2 \rightarrow 2NO_2^- + 4H^+ + 2H_2O$$

Nitrobacter

$$2NO_2^- + O_2 \rightarrow 2NO_3^-$$

Total reaction

$$NH_4^+ + 2O_2 \rightarrow NO_3^- + 2H^+ + H_2O$$

When including nitrogen used for cell synthesis, Randall et al. (1992) showed an oxygen requirement of 4.3 g O_2/g-NH_4-N oxidized to nitrite, and a cell yield of 0.15g VSS/g NH_4-N oxidized and 0.02 g VSS/g NO_2-N oxidized. Approximately 7.14 mg of alkalinity as $CaCO_3$ is consumed per mg NH_4-N oxidized. The U.S. EPA Nitrogen Control Manual (1993) uses an oxygen requirement of 4.6 g O_2/g NH_4-N for the oxidation of ammonia to nitrate, a yield of 0.10g VSS/g NH_4-N oxidized, and estimates 7.1 mg of alkalinity as $CaCO_3$ is consumed per mg NH_4-N oxidized. Metcalf & Eddy (1991) also reports an oxygen requirement of 4.3 g/g-N for the oxidation of ammonia to nitrate, a typical design yield of 0.20g VSS/g NH_4-N oxidized, and uses a design rate of approximately 8.64 mg of alkalinity as HCO_3 is consumed per mg NH_4-N oxidized.

In active nitrifying systems *Nitrobacter* nitrite oxidation rates are much higher than *Nitrosomonas* ammonia oxidation rates (Stensel, 1990) and the kinetics of nitrification are generally based on the NH_4-N oxidation rates. The nitrifyer growth rate can be modeled using the equation below (EPA, 1993):

$$\mu_N = \mu_{M,MAX} \times \left(\frac{N}{K_N + N} \right) \qquad (2.1)$$

Where μ_N = specific growth rate of *Nitrosomonas* per day

$\mu_{M,MAX}$ = maximum specific growth rate of *Nitrosomonas* per day

K_N = half-saturation coefficient for *Nitrosomonas*, mg/L of NH_4-N

N = NH_4-N concentration, mg/L

Values for $\mu_{N,max}$ range from 0.84 to 1.32 d^{-1} and 0.6 to 3.6 mg/L for K_N at 20 °C (EPA, 1993). The maximum specific growth rates for these two nitrifying bacteria can be greatly affected by temperature, pH, and dissolved oxygen (DO) levels. The nitrification specific growth may decrease about 50% when reactor temperature drops from 20° C to 12 °C. Maximum rates of nitrification have been observed between pH values of about 7.2 and 8.0. A 50% decrease in nitrifying bacteria growth rates is also observed as system pH drops below 6.5. Nitrifying bacterial growth rates are also strongly influenced by DO levels with an estimated 50% reduction in growth rates at DO concentrations of 1.3 mg/L, versus an estimated 25 % reduction in nitrifyer growth rates at DO concentrations of 3.9 mg/L. Nitrifying bacteria are sensitive organisms and are extremely susceptible to a wide variety of inhibitors. A variety of organic and inorganic agents can inhibit growth of the nitrifying bacteria. High concentrations of ammonia and nitrous acid can also be inhibitory to nitrification process (Metcalf & Eddy, 1991).

2.5 Denitrification Fundamentals

Nitrate reduction in wastewater systems occurs through assimilation and denitrification. During cell synthesis some of the nitrate is reduced to ammonia and assimilated into cell as discussed earlier. During the denitrification process nitrate is reduced to nitrite, to

nitric oxide, to nitrous oxide, and to nitrogen in a four-step process (Payne et al, 1981):

$$NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2$$

During the denitrification process nitrate in the wastewater is reduced by many organisms that use nitrate as electron acceptors for energy metabolism. The use of nitrate for energy metabolism is referred to as dissimilative metabolism. Whereas many organisms carry out dissimilative metabolism of nitrogen, only a restricted variety of organisms carry out assimilative metabolism. A comparison of assimilative and dissimilative processes for the reduction of nitrate is shown in Figure 2.4.

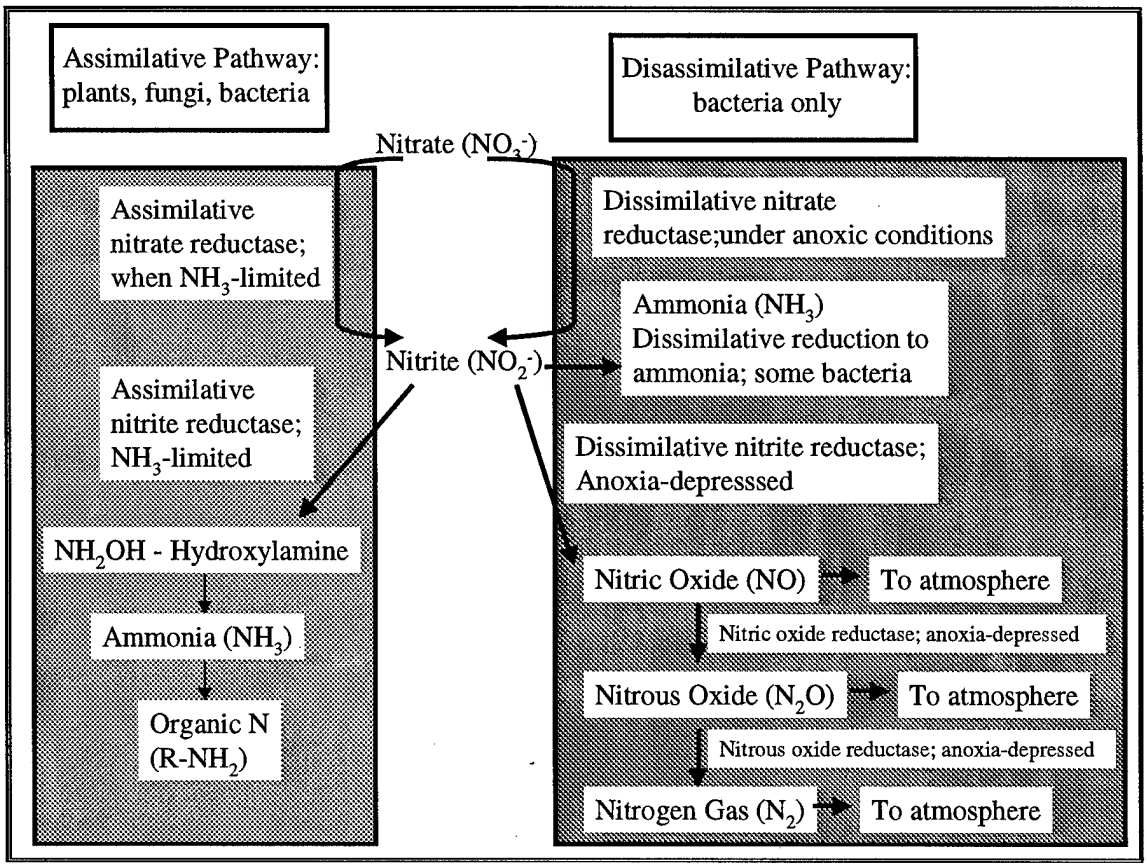

Figure 2.4 Assimilative and Dissimilative Processes for Reduction of Nitrate (Brock, 1991)

During assimilatory nitrate reduction, nitrate is reduced to the oxidation level of ammonia for use as a nitrogen source for growth. During dissimilatory nitrate reduction, nitrate is used as an electron acceptor in oxidation-reduction energy generating reactions. The enzyme first involved in nitrate reduction is *nitrate reductase*, a molybdenum-containing enzyme. These enzymes are generally soluble proteins that are oxygen repressed and synthesized under anaerobic conditions. Because O2 inhibits synthesis of the dissimilative nitrate reductase, the denitrification process is considered a strictly anaerobic process. The effects of DO are discussed in detail in section 2.5.1.

The first product of nitrate reduction is nitrite and another enzyme, *nitrite reductase* is responsible for the next step in the denitrification process. From nitrite two routes of reduction are possible, either to ammonia or directly to nitrogen gas. A large number of bacteria can reduce nitrite to ammonia for cell synthesis. However, unless ammonia is limiting, this process is not significant in wastewater treatment. There are some bacteria that can reduce nitrite directly to nitrogen gas through two intermediate steps. Several bacteria are known only to reduce nitric oxide to nitrous oxide. In any case, the completion of the reduction of nitrite to nitrogen gas requires the *nitric oxide reductase* enzyme to reduce nitrite to nitric oxide and the *nitrous oxide reductase* enzyme to reduce nitric oxide to nitrogen gas. The various types of bacteria found in wastewater responsible for ultimately reducing nitrate to nitrogen gas are discussed next.

Many different and very diverse facultative heterotrophic bacteria are known to carry out denitrification. Denitrifiers are ubiquitous in most municipal wastewaters and sludges (Henze et al. 1987). Some of the heterotrophic genera are *Achromobacter*, *Acinetobacter*, *Agrobacterium*, *Alcaligenes*, *Arthrobacter*, *Bacillus*, *Morazella*, *Neisseria*, *Paracoccus*, *Pseudomonas*, *Rhizobium*, *Spirillum*, and *Vibrio*.

Pseudomonas species are the most common and widely found denitrifying bacteria. These bacteria have been shown to use a wide range of organic substrates such as hydrogen, methanol, organic acids, and other aromatic compounds (Payne, 1981). Some

species begin in the denitrification process by reducing nitrite and not nitrate. Some species accumulate nitrous oxide instead of nitrogen gas as their terminal end product Randall et al. (1992).

Bacillus and *Flavobacterium* are also commonly found denitrifying bacteria. Some *Bacillus* species can reduce nitrate and nitric acid but not nitrite or nitrous oxide. These bacteria are also capable of using wide range of organic substrates. *Flavobacterium* reduce nitrite and nitrous oxide to nitrogen gas, but cannot reduce nitrate. They can use only use simple carbohydrates for carbon sources.

The following subsections address the specific effects of dissolved oxygen, pH, wastewater characteristics, electron acceptors, temperature, and substrate addition on the denitrification process.

2.5.1 Effect of Dissolved Oxygen (DO) on Denitrification

As discussed in the previous section, the presence of oxygen suppresses the production of the nitrate, nitrite, nitric oxide, and nitrous oxide reductase enzymes needed for denitrification. DO concentrations as low as 0.2 mg/L have been reported to inhibit denitrification by a *Pseudomonas* culture (Teri and Mori, 1975) and stop denitrification in pure cultures (Grau et al, 1982). DO concentrations as low as 0.2 mg/L have also been reported to inhibit denitrification within activated sludge treatment (Dawson and Murphy, 1972). Henze et al (1991) reported complete denitrification inhibition at DO concentrations varying from 0.3 to 1.5 mg/L. Both Wheatland et al. (1959) and Batchelor et al. (1982) found that denitrification rates were maximized at zero DO levels and that rates decreased almost exponentially with increasing DO levels. At DO levels less than 1 mg/L, the observed specific denitrification rates were generally a linear function of the DO concentration as represented in Equation 2.2 (Crites, 1998):

$$SDNR_{obs} = SDNR_{T20} \text{ (1-DO) where DO is in mg/L} \qquad (2.2)$$

At low DO concentrations in activated sludge the inner portion of the activated sludge floc particles may be at zero DO concentration creating anoxic microcosms. When anoxic microcosms are established at low DO, denitrification can occur within the floc while nitrification is occurring on the outside of the floc. Stensel et al (1995) observed denitrification in an oxidation ditch at DO levels between 1.0 and 2.5 mg/l. Quantifying the effects of DO concentration on denitrification rates within activated sludge systems is complex because many factors affect the anoxic portion of the floc and the nitrate reduction rate within the floc. These factors include floc size, floc density, organic substrate concentration, and diffusion limitations of the microbial floc (Grau et al., 1984). With larger floc sizes and higher bulk liquid substrate concentrations a greater portion of the floc would be anoxic and thus allow more nitrate reduction.

The transition from aerobic respiration to denitrification is not necessarily a spontaneous process. After extended periods of aeration, the literature indicates that time is required for the denitrifiers to activate their nitrate reductase enzyme. Payne and Riley (1969) showed that 40 minutes was required for an aerobically grown culture to shift to maximum activity under anoxic conditions. Payne and Riley also noted that once denitrification enzymes were produced, they continued to function for 'some time' after an aerobic environment was re-established. Ekama, Dold and Marais (1986) also noted that activated sludge from anoxic aerobic systems required an acclimatization period to anoxic conditions following extended aeration periods.

2.5.2 Effects of pH on Denitrification

In general, pH influences are less of a concern in denitrification than for nitrification. Nitrification consumes alkalinity at 8.64 mg HCO_3^-/mg NH_4-N oxidized while denitrification produces alkalinity. In heterotrophic denitrification reactions, one equivalent of alkalinity is produced per equivalent of nitrate-nitrogen reduced. This equates to 3.57 mg of alkalinity as $CaCO_3$ / mg NO_3-N, or about one-half the amount

destroyed through nitrification Randall et al. (1992). Optimum pH ranges of 6.5 to 7.5 were reported for denitrifying bacteria with an optimum condition around 7.0 (Metcalf & Eddy (1991) and Crites (1998)). Urbain et al (1997) reported that a pH of 7.5 was optimum and a pH higher or lower than this led to decreased denitrifying activity. Urbain also reported observing a linear decrease in denitrification rates as the pH was increased from 8.0 to 9.5 and decreased from 7.0 to 4.0.

2.5.3 Wastewater Characteristics and Denitrification

The EPA Nitrogen Control Manual (EPA, 1993) states that the nature of carbonaceous material is a very important process variable in denitrification processes. The manual points out that soluble organics are utilized much more readily by the heterotrophic biomass for denitrification than less biodegradable particulate and colloidal forms of carbon. Research by Garber et al (1994) and others has also supported the observation that the soluble COD is consumed during denitrification. With the use of more complex processes such as BNR and biological phosphorus removal, wastewater treatment plant designers have recognized that specific wastewater characteristics are very important and that the fraction of soluble degradable COD in the influent wastewater COD can have a major effect on system performance. Furthermore, the wastewater characteristics vary for different locations and should be determined and incorporated in process design.

In modeling municipal activated sludge processes Henze et al. (1987) and Clayton et al. (1991) advocate dividing the wastewater COD into two fractions, a readily biodegradable chemical oxygen demand (RBCOD) and a slowly biodegradable chemical oxygen demand (SBCOD). RBCOD is assumed to be made up of dissolved organic molecules that easily pass through the cell wall, while the SBCOD is assumed to consist of colloidal or larger complex molecules that cannot pass directly through the cell wall and require hydrolysis by extracellular enzymes. The SBCOD is believed to be adsorbed and stored on the organism where it is hydrolyzed by surface attached extra-cellular enzymes and then transferred directly through the cell wall for utilization (Clayton et al., 1980). The

IAWPRC Task Group for activated sludge modeling assumes the SBCOD is converted to RBCOD in the liquid phase, which is then utilized at a rate according to Monod's equation (Dold and Marais, 1986).

Mamais et al. (1993) described a measurement technique for quantifying the wastewater RBCOD fraction and non-biodegradable or inert wastewater SCOD in municipal wastewater. The RBCOD is assumed to represent a 'truly' soluble organic matter that is determined by 0.45μ membrane filtration of a chemically flocculated sample. A zinc hydroxide precipitate captures colloidal particles during flocculation so that only COD consisting of low molecular weight molecules and truly dissolved COD pass through the filter membrane. The COD (SCOD) for the filtered sample is determined which consists of the dissolved organics. A portion of the SCOD may not be biodegradable and the non-biodegradable SCOD (NBSCOD) must be subtracted from the SCOD to yield the RBCOD. Ekama et al. (1984) reported that activated sludge systems with an MCRT of 3 days or greater produce an effluent soluble COD that is mostly non-biodegradable. Thus the filtered SCOD is corrected to obtain the RBCOD by subtracting the flocculated SCOD concentration determined from the activated sludge effluent wastewater at an MCRT of 3 days or more. The slowly biodegradable COD (SBCOD) for a wastewater sample is the soluble COD (0.45μ filtration) minus both the RBCOD and NBSCOD. NBSCOD is the COD remaining after membrane filtration of an activated sludge treated effluent sample.

Ekama et al. (1986) related denitrification rates to readily and slowly biodegradable SCOD fractions in wastewater. They studied nitrate removal rates in anoxic batch tests. In these tests wastewater, activated sludge, and nitrate were added to a reactor and continuously mixed. The nitrate concentration was measured over a period of four to five hours and initially the concentration decreased at a rapid constant rate. They claimed that this rate represented utilization of the readily biodegradable COD fraction of the wastewater. A slower and constant nitrate utilization rate followed the rapid removal rate and was claimed to be caused by the utilization of particulate and colloidal biodegradable

COD. Ekama developed nitrate utilization rates based upon the COD fractions. Ekama used these results to show how COD fractions are important in determining denitrification capacity and detention times for a given batch reactor volume.

Ekama et al. (1986) also provided general values for unbiodegradable COD fractions in municipal wastewater. For the particulate unbiodegradable COD fraction, a value of 0.13 was given for unsettled municipal wastewater and 0.04 for settled municipal wastewater. For the soluble unbiodegradable COD fraction, a value of 0.05 was given for unsettled municipal wastewater and 0.08 for settled municipal wastewater. The resulting unbiodegradable COD (UBCOD) fraction for municipal wastewater is 0.18 for unsettled wastewater and 0.12 for settled wastewater. A small fraction of the UBCOD passes through a WWTP and leaves as part of the effluent. The major portion of UBCOD becomes enmeshed within the floc and is removed with the floc through the wasting of the activated sludge. The enmeshed portion of the UBCOD directly effects the active biomass.

The active biomass is considered the portion of the activated sludge that is biologically active or capable utilizing nutrients during the oxidation of the carbonaceous material within the wastewater. Variations in unbiodegradable COD fractions between different wastewaters have a strong effect on the active biomass fraction and substrate utilization rates in activated sludge. Higher wastewater UBCOD fractions generally result in smaller active biomass fractions. Since most specific nutrient removal rates are reported in grams of nutrient removed per gram of MLVSS, smaller active biomasses result in reduced specific utilization rates (SUR). SURs reported in terms of active mass provide more precise and more comparable utilization rates.

2.5.4 Effect of Electron Acceptor on Substrate Utilization Rates

Laboratory and field measurements have shown that substrate utilization rates decrease when nitrate is used as the electron acceptor instead of oxygen. This difference has been

attributed to the possibility that only a portion of the heterotrophic population is facultative bacteria. This difference in aerobic and anoxic reaction rates is accounted for in the IAWQ's ASM2 model by a term, η_g, which is the fraction of active biomass responsible for substrate utilization under anoxic conditions. The default value for η_g in the ASM2 model is 0.37.

2.5.5 Temperature

The literature is in general agreement that temperature affects denitrification rates, but the degree to which temperature affects denitrification rates varies within the literature. The impact of temperature on denitrification rates is most often described by an Arrenhius-type function:

$$SDNR_T = (SDNR_{20})\theta^{T-20} \qquad (2.3)$$

Where θ = temperature coefficient

Table 2.2 below lists reported θ coefficients for correcting denitrification rates in municipal wastewater treatment.

Table 2.2 Temperature Coefficients Reported for Denitrification

Reference	θ	*Fraction of 20 °C rate at 10 °C*
Ekama and Marais et al 1984	1.20	0.16
Barnard et al 1993	1.09	0.42
Dawson and Murphy et al 1972	1.06	0.56
Metcalf & Eddy (1991)	1.09	0.42
Crites (1998)	1.09	0.42
IAWQ Model – BioWin (1995)	1.029	0.75

The reported θ values vary from 1.03 to 1.20 and can result in the prediction of denitrification rates at 10 °C that are from 16 to 75% of the predicted 20 °C rate. The

U.S. EPA (1993) lists temperature correction coefficients for modeling denitrification ranging from 1.03 to 1.20.

The temperature effects on denitrification rates may be further complicated as some research has indicated that different temperature coefficients apply to SDNRs associated with the different COD fractions. Clayton et al. (1991) studied the relationship between COD fractions and denitrification at a WWTP using Bardenpho process. He reported three rate functions for denitrification as a function of different COD fractions in the wastewater and temperature (Table 2.3):

Table 2.3 SDNRs as a Function of COD Fraction and Temperature

Substrate	θ	*SDNR*
RBCOD	1.20	$0.720*\theta^{T-20}$
First Anoxic SBCOD	1.08	$0.101*\theta^{T-20}$
Second Anoxic SBCOD	1.029	$0.072*\theta^{T-20}$

His SDNRs were based on a first order rate model with respect to the active VSS concentration. The SDNR for each substrate is considered to be zero order with respect to both nitrate and substrate concentrations. The highest and most temperature sensitive SDNRs were observed in the first anoxic zone with RBCOD utilization. The lowest and least temperature sensitive SDNRs were observed in the second anoxic zone with SBCOD utilization.

2.5.6 COD Consumption During Denitrification

The following shows oxidation-reduction reactions for using acetic acid or raw sewage as substrates for denitrification (Frick and Richard, 1985):

Acetic Acid

$$5\ CH_3COOH + 8\ NO_3^- \rightarrow 4\ N_2 + 10\ CO_2 + 6\ H_20 + 8\ OH^-$$

Sewage

$$C_{10}H_{19}O_3N + 10\ NO_3^- \rightarrow 5\ N_2 + 10\ CO_2 + 3\ H_20 + NH3 + 10\ OH^-$$

The COD per mole acetate and sewage is 76.8 and 464 mg/L, respectively. These equations then indicate that the COD :NO_3-N ratios are 2.08 and 3.08 for acetate and sewage, respectively. Barth et al. (1968) estimated the ratio of BOD:NO_3-N consumption as 4.0. Randall et al. (1992) obtained a COD:NO_3-N ratio of 3.45 in laboratory studies and noted the ratio of COD:NO_3-N consumption is related to the net cell yield.
The mass of COD oxidized through denitrification can be accounted for by estimating the equivalent amount of oxygen provided when nitrate is used as the electron acceptor. Barker et al. (1995) showed that the net half reaction for the four-step process of denitrification is as follows:

Reaction for each step :

$$2e^- + NO_3^- + 2H^+ \rightarrow NO_2^- + H_2O$$

$$e^- + NO_2^- + 2H^+ \rightarrow NO + H_2O$$

$$2e^- + 2NO + 2H^+ \rightarrow N_2O + H_2O$$

$$2e^- + N_2O + 2H^+ \rightarrow N_2 + H_2O$$

Net reaction:

$$10e^- + 2NO_3^- + 12H^+ \rightarrow N_2 + 6H_2O$$

The Half Reaction for 1 mole of electron transfer is:

$$e^- + 1/5NO_3^- + 6/5H^+ \rightarrow 1/10N_2 + 3/5H_2O$$

The Half Reaction for the reduction of oxygen is:

$$e^- + 1/4O_2 + H^+ \rightarrow 1/5H_2O$$

The two half reaction show that the oxidation of one electron requires the reduction of ¼ mol of oxygen or 1/5 mol of nitrate-nitrogen. Using the molar relationship between oxygen and nitrate and their molecular weight shows that the reduction of 1 mg NO_3-N is equivalent to the same amount of electron transfer as 2.86 mg oxygen.

Stensel et al. (1990) used the oxygen equivalent of nitrate nitrogen in oxidation-reduction reactions and net cell yield to estimate COD:NO_3-N ratio. He assumed that the COD of

biomass was 1.42 g COD/g VSS and that the VSS contained 10% nitrogen. Using these assumptions the COD: N consumption ratio for denitrification is shown in the Equation below:

$$COD / N = 2.86 \ (1\text{-}1.134 * Y_n) \qquad (2.4)$$

where Y_n = net biomass synthesis yield coefficient(g VSS/g COD used)

Urbain et al. (1997) found in lab scale batch reactors that a C/N ratio of 1.3 and greater allowed for complete nitrate removal at 25 °C with an influent composed of 20% VFAs. They concluded from their research that a C/N ratio of 1.4 was optimum for complete denitrification within batch reactors.

Reasonably accurate mass balances are needed to determine COD/NO_3-N consumption ratios for laboratory or full-scale denitrification processes. These require a COD and nitrogen balance on influent and effluent composite samples and a complete and accurate inventory of solids in the system and solids wasted as well as the nitrogen and COD content of the solids (Barker et al., 1995). 'Steady state' operating conditions are preferred for mass balance analyses.

An effective dosage of methanol (mg methanol per mg of nitrate reduced) has been reported as 2.55 mg of methanol/mg of nitrate reduced (Bradstreet et al., 1994). This is consistent with the ratio of 2.47 mg of methanol/mg of nitrate reduced in the EPA manual (EPA, 1993).

2.6 Specific Denitrification Rates (SDNR)

A commonly used and convenient empirical design parameter used to size anoxic tanks has been the specific denitrification rate (SDNR), which is the g NO_3-N reduced/g MLVSS-day. This value has been used to size anoxic tank volumes, based on the amount of nitrate to be removed and the tank MLVSS concentration:

$$V = \Delta NO_3\text{-}N / (SNDR)*X \qquad (2.5)$$

Where:

V = Anoxic tank volume, m3

ΔNO_3-N = Amount of NO_3-N to be removed, g/day

SDNR = Specific denitrification rate, g NO_3-N reduced/g MLVSS-day

X = MLVSS concentration, g/L

SDNR rates were observed from bench scale; pilot scale and full-scale plant tests and Burdick et al. (1982) noted that the SDNR was directly proportional to the applied food to microorganisms' (F/M) ratio to the anoxic reactor. The design SDNR could be estimated from the F/M ratio using Equation 2.6:

$$SDNR = 0.03*F/M + 0.029 \qquad (2.6)$$

SDNRs by various investigators are summarized in Table 2.4 for municipal wastewater treatment systems. A wide range of SDNR values are shown.

Table 2.4 Reported SDNRs for Anoxic Zones in Municipal Wastewater Treatment Processes

Source	*SDNR (g NO_3-N/gMLVSS-d)*	*Systems*
Henze et al. (1991)	0.024 –0.168	Observed at full scale plants
Murakami et al. (1998)*	0.053 – 0.42	Bench scale anoxic selector
Bradstreet et al. (1994)	0.015 – 0.062	Full scale A/O process
Reardon et al. (1996)	0.032 – 0.07	Batch tests at LOTT WWTP
Metcalf & Eddy (1991)	0.03 - 0.11	Observed Design Values for 1st Anoxic Zone
Ekama et al. (1986)*	0.072 - 0.72	Anoxic Batch Tests
Burdick et al. (1982)	0.035 0.035 – 0.08	Bardenpho Process Laboratory and Pilot Plants
Hong et al. (1997)	0.008 - 0.02	Bench scale A/O and Bardenpho
Hong et al. (1997)	0.027 – 0.035	Bench scale A/O and Bardenpho
EPA (1993)	0.04	Recommended Design Value for 1st Anoxic Zone

*Rates reported by Murakami and Hong were in terms of MLSS and not MLVSS. Rates reported by Ekama were in terms of active VSS, not MLVSS.

Murakami et al. (1998) reported that the lowest SDNRs reflected high mixing rates and the highest SDNRs was with low mixing and short anoxic detention time in a bench scale anoxic zone receiving primary effluent and RAS. Additionally, the first anoxic zone bench tests resulted in reported SDNR values of 0.053 – 0.100 g NO_3-N/gMLVSS-d for a hydraulic detention time of 35 minutes and varying the reactor mixing speed from 200 rpm (0.053) to 50 rpm (0.100). When bench tests were repeated using a constant mixing speed of 50 rpm and varying anoxic detention times from 17 to 164 minutes, SDNR rates ranged from 0.416 g NO_3-N/gMLVSS-d (17 min DT) to 0.086 g NO_3-N/gMLVSS-d (164 min DT). Bradstreet et al. (1994) used a full-scale operation study at a WWTP using an A/O process to determine the response of the process to varying operational conditions. The SDNR rates were observed over an eight month period and reflect temperatures ranging from 9.7 to 22 °C. The highest SDNR was actually observed in December when temperatures were 14 °C. The rates reported by Metcalf & Eddy (1991) are typical denitrification rates for wastewater at temperatures between 15-27 °C and were not associated with any particular type of treatment process.

Ekama et al. (1986) reported SDNRs ranging from 0.72 to 0.072 g/g-d, as soluble substrate was depleted in batch denitrification rate tests. In these batch tests, Ekama observed three sequential linear denitrification rates. The first and higher rate reflected denitrification rates using RBCOD as a substrate and the second slower rate reflected denitrification rates using SBCOD as the primary substrate. The last rate reflected denitrification due to endogenous activity. Burdick et al. (1982) compared the denitrification rates of determined in the first anoxic stage of a Bardenpho system with a weak wastewater to SDNRs determined by previous work as a function of anoxic zone F/M ratio to develop Equation 2.6.

The observed SDNR rates in Table 2.4 are quite variable and reflect differences in wastewater characteristic, DO concentrations, pretreatment processes, temperature, anoxic reactor detention time, fraction of active mass in the MLVSS, and the F/M ratios of the anoxic reactors studied. For example, if primary treatment is used the MLVSS concentration in the anoxic tank should have a higher fraction of active biomass than for a system with no primary treatment. With primary treatment less inert non-biodegradable solids are fed to the secondary treatment system so that a greater proportion of the MLVSS developed is due to biomass. Since SDNR is based on MLVSS, a higher fraction of active biomass will result in a higher SDNR value. Since RBCOD results in a faster biodegradation rate then that from SBCOD, wastewater with a higher fraction of RBCOD should provide higher observed SDNRs. SDNR values taken from tests on anoxic tanks with longer detention times should also be lower since lower concentrations of degradable COD and nitrate could exist. The longer anoxic tank detention time could provide a lower F/M ratio that according to Equation 2.6 would result in a lower SDNR.

Many treatment plants also use external carbon sources to increase activated sludge system denitrification rates in anoxic tanks after nitrification, to increase rates in MLE anoxic zones for wastewaters with low RBCOD concentrations, or to increase rates for anoxic zones operating at low temperatures. Reported SDNR values for systems using external COD addition also vary widely as shown in Table 2.5 below:

Table 2.5 Reported SDNRs for Anoxic Reactors Using External Substrate Addition

Source	*Substrate Added*	*SDNR* (gNO_3-$N/gVSS$-d)	*Temperature* (^oC)
Henze et al. (1989a)	Acetic Acid/ Methanol	0.48	
Barber et al. (1994)	Acetic Acid/ Methanol	0.7-1.6	35-38
Metcalf and Eddy (1991)	Methanol	0.21-0.32 0.12-0.20	25 20
Urbain et al. (1997)	Methanol	0.28 ± 0.07	25
Urbain et al. (1997)	Fermentation supernant	0.39 ± 0.02	25
EPA (1993)	Methanol	0.25	

The SDNRs reported by Barber et al. (1994) were for an industrial WWTP operating at 35-38 °C with methanol and acetate addition. Henze et al. (1990) reported that acetate or methanol addition produces maximum SDNRs in BNR processes. Henze reported that a typical observed SDNR with methanol and acetate is 0.48 g NO_3-N/gMLVSS-d compared to a maximum observed SDNR of 0.168-g NO_3-N/gMLVSS-d in full-scale denitrification systems using municipal wastewater only. However, he also reported that carbon source addition would also increase overall sludge production by 10-20 percent depending on the amount of additional carbon added. Urbain et al. (1997) explored the effects of methanol addition versus fermentation tank supernant addition on nitrogen removal in a full-scale plant. He found that the fermentation tank provided a better carbon source in terms of removal rates.

Potter et al. (1998) used a bench scale prototype of a partial nitrification/complete denitrification (PN/CD) system to observe nitrogen removal performance of an aerobic-anoxic system with recycle from the anoxic zone operated so that no nitrate breakthrough was observed in the prototype's effluent. A computer model was used compare the nitrogen removal performance of the PN/CD system to that of a conventional MLE process producing the same quality effluent. The simulation results indicated less energy was required for mixed liquor recirculation and recycle resulting in a 26 to 44% smaller total reactor volume compared to an MLE process. However, the majority of the total volume reduction was due to aerobic zone sizing and the design focuses minimizing effluent nitrate concentrations and not on total nitrogen removal, allowing effluent ammonia concentrations of 10 mg/L or greater to be observed throughout the prototype operation and model implementation.

Zhao, Mavinic, and Oldham (1996) reported that methanol addition improved denitrification rates in the anoxic zone of Bardenpho process at all dosages used (15 to 60 mgCOD/L). During both lab and field experiments they noted a lag period between methanol addition and an observed increase in both nitrogen removal and denitrification rates of 1 to 2 days. Bradstreet et al. (1994) explored the effects of adding 30 mg/L of

methanol to the anoxic zone of an A/O process and noted a one to day lag period before a change in nitrate removal was noted and five days before significant nitrate removal was experienced. After five days of addition the effluent nitrate levels had been reduced from 7.8 mg/L to 4.3 mg/L.

Actual anoxic zone sizes and HRTs can vary widely between WWTPs. Some reported anoxic zone HRTs are shown in Table 5.6.

Table 2.6 Reported WWTP Anoxic Zone HRTs

Reference	*WWTP Type*	*HRT (Hours)*
Burdick et al. (1982)	Bardenpho-1st Anoxic	2.7
Burdick et al. (1982)	Bardenpho-2nd Anoxic	2.2
Fries et al. (1994)	Modified Bardenpho –1st /2nd Anoxic	.2
Jain et al. (1992)	A/O Process	.28

Fries et al. (1994) used a modified Bardenpho process with anoxic, anaerobic, anoxic, aerobic treatment with fermentation tank addition to the anaerobic zone. The EPA manual (EPA, 1993) recommends anoxic detention times of 1 to 3 hours

2.7 Endogenous Denitrification

Denitrification does occur under endogenous conditions within activated sludge systems. Release of organic carbon during cell death and auto-oxidation by living cells creates an oxygen or nitrate demand. Reported endogenous SDNR rates are shown in Table 2.7 below.

Table 2.7 Reported Endogenous SNDR Values

Source	*Rate* (gNO_3-N/g VSS-d)	*Plant Type*
Henze et al (1990)	0.0048- 0.012	Observed at full scale plants
Hong et al (1997)	0.0082	Bench scale A/O and Bardenpho
Stensel et al (1995)	0.005 – 0.015	On/Off Oxidation Ditch
Ekama et al (1984)	0.072	Batch anoxic reactor
Randall (1992)	0.04 – 0.05	Design value

Burdick et al. (1982) indicated that the SDNR in the second anoxic stage of a Bardenpho process is dependent on the endogenous respiration rate of the biological solids and is a function of SRT. They compared the specific denitrification rate determined in the second anoxic zone of a Bardenpho process with previous work in which domestic water was treated without primary treatment and found that the SDNR can be also estimated using the SRT using Equation 2.6 below:

$$SDNR = 0.12 \times \theta^{-0.706} \qquad (2.7)$$

The endogenous SDNR value should be related to endogenous respiration rate of the mixed liquor. Table 2.7 shows typical endogenous decay rates for activated sludge.

Table 2.8 Reported Endogenous Decay Design Values

Source	*Rate* (g biomass /g biomass-d)
Randall et al. (1992)	0.04 – 0.05
BioWin Model (1995)	0.08
Metcalf & Eddy	0.017 – 0.048

Using the endogenous decay values in Table 2.7 the endogenous SDNRs were estimated assuming 1.42 g COD/g VSS, 2.86 g O_2/g NO_3-N, 0.8 g O_2/gCOD, and 50% of the active mass capable of using nitrate instead of oxygen. The resulting SDNRs range from 0.007

to 0.032 gNO_3-N/g VSS-d. These rates are fairly close to the range of SDNR reported in the literature and shown in Table 2.6 (0.0048 to 0.072 gNO_3-N/g VSS-d).

Ekama, Dold, and Marais (1986) conducted tests to determine the average daily endogenous respiration rate based on active biomass volatile suspended solids (AVSS) and the effects of temperature rate. They found endogenous respiration to be approximately 0.24 mg AVSS/mg AVSS-day and independent of sludge age between sludge ages from 2.5 to 20 days when temperature was a constant 20 °C. They also found that endogenous respiration was not very temperature sensitive and used the following equation to correction endogenous respiration for temperature:

$$b_{h\text{-}T20} = b_h*(1.029)^{T-20} \qquad (2.8)$$

This equation is the same temperature correction factor used for growth with COD under anoxic and aerobic conditions in the IAWQ model.

2.8 Oxidation-Reduction Potential (ORP) Relationships to Denitrification Processes

ORP controlled aeration is often used to control the denitrification process within wastewater treatment plants. This is especially true for plants not originally designed for nitrogen removal. ORP is a measure of the activity of electrons involved in the oxidation-reduction reactions within an aqueous environment. The observed ORP represents the net electron activity of all the redox reactions taking place in aqueous solution and is direct indicator of the general oxidative status of the system (Kjaergaard (1977); Koch et al. (1985); Stensel et al. (1995)). ORP values can be measured over the full range of redox conditions. In most complex biological systems, such as wastewater treatment applications, the many chemical and biological oxidation-reduction reactions taking place are not in equilibrium and the observed ORP cannot be interpreted thermodynamically (Kjaergaard (1977)). However, ORP values do reflect all factors that contribute to the electron activity in solution such as the chemical constituents of the system, variety of biological activity, pH, and temperature.

ORP values are specific to the system in which they are measured. Real time control research with wastewater processes has shown that absolute ORP values from different probes in the same sewage have been observed to vary up to 50 mV, while simultaneously showing identical relative changes with time (Wareham et al., 1992). Researchers have found that the ORP varies linearly with the log of the oxygen concentration, indicating that ORP is a sensitive parameter at very low oxygen levels (Ishizaki 1974, Radja 1984, and Shibia 1974). ORP measurements are considered a more precise measurement of the extent of a systems oxidative conditions, since DO measurements are not reliable at low DO levels and are meaningless at zero DO concentrations.

Alternating aeration/mixing was instituted at an oxidation ditch in Eastern Washington to incorporate nitrogen removal and help control SVI's by Stensel et al. (1995). One anoxic cycle per day was used at the WWPTP to explore nitrogen removal results at different cycle lengths. ORP was measured and used to determine when nitrate was consumed within the oxidation ditch. When nitrate was nearly fully depleted the ORP value dropped dramatically. Using ORP controlled denitrification and one anoxic cycle per day the plant was able to achieve effluent total nitrogen concentrations of less than 10.0 mg/L. The use of additional cycles resulted in increased nitrogen removal rates. Additional denitrification within the oxidation ditch was observed during aerobic cycles of operation as well.

Lo et al. (1994) also examined the use of ORP to control nitrogen removal. In their experiments they used a synthetic sewage with total nitrogen (TN) concentrations ranging from 34 to 44 mg/L, an MLSS of 2900-4100 mg/L, a F/M ratio of .11 to .13 g COD/gMLSS-d, a recycle ratio of 0.5 to 1.5, an HRT of 20 hours, and a sludge age of 20 days. These parameters were used to evaluate TN removal within two different system configurations.

In the first system configuration, a 90-liter ORP controlled aerobic tank with ORP set points of 40, 70, and 110 mV was used. They found a 70-mV set point provided the highest TKN removal rate (96%). The 40 mV set point kept DO levels <0.2 mg/L, but nitrification was limited resulting high tank effluent ammonium levels. A 100-mV set point on the other hand resulted in complete nitrification with limited denitrification. They found that equal 15-minute aeration and mixing conditions resulted in DO ranges between <0.1 mg/L and approximately 0.5 mg/L resulted in a favorable balance between nitrification and denitrification. Phosphorus removal was relatively constant at all set points.

In the second system configuration, Lo et al. (1994) used a small anoxic selector (3.8 L), followed by ORP controlled aeration (72 L), and ending with a small aerobic zone (18L). ORP set points of 70, 110, and 180 mV were used for this system. The 110-mV setting provided the best TN removal rate (93%). For this system, the 110-mV set point produced the same range of DO levels as the first system with the 70-mV set point. The difference in removal rates was attributed to a higher TN level used in the second system (33 mg/L versus 44 mg/L) and differences in ORP zone sizes. Enhanced phosphorus removal was also indicated in the second system by both higher levels of phosphorus removal and higher biomass phosphorus content.

Lo et al. (1994) concluded that a small anoxic selector could either control or cure bulking problems in activated sludge processes. The authors concluded that an equal distribution between aeration and non-aeration periods with a maximum DO level < 0.5 mg/L in the aeration cycle provided optimum TN removal from their laboratory studies. ORP control appears to be viable method for obtaining nitrogen removal within both existing plants not designed for nutrient removal and existing plants with limited anoxic zones and large aerobic zones.

2.10 Biological Phosphorus Removal and Denitrification

Biological Phosphorus Removal (BPR) processes are often found in conjunction with BNR processes. When BPR is incorporated into a BNR process such as the MLE or Bardenpho process, an anaerobic contact zone is added before the anoxic zone. The reason for this is that a prerequisite for BPR is the availability of short chain VFAs such as acetate. BPR processes select for the growth of phosphorus accumulating organisms (PAOs) which can store acetate under anaerobic conditions. The acetate is converted to PHB within the PAOs. Under oxic conditions the PAO oxidize the PHB and grow along with the other heterotrophic bacteria. The PAOs exhibit lower yields than non-PAO bacteria. Without the competitive advantage of being able to store VFAs in the form of PHB under anaerobic conditions, the normal heterotrophs with much higher yields would out grow the PAOs and dominate the biomass within the activated sludge and effectively 'wash them out'. A high PAO content within the activated sludge is desired. The PAOs have an extremely high phosphorus content within their cells and when solids are wasted each day the phosphorus is removed as part of PAO cell mass. The BPR process is believed to have two different impacts on the specific denitrification rate within the anoxic zone based on whether or not the PAO population can use nitrate as an electron acceptor.

First, if PAOs are strict aerobes, they do not use nitrate as an electron acceptor and thus do not mediate the denitrification processes. A higher aerobic only PAO fraction within the active biomass would result in less organic substrate available for denitrification since the PAOs can assimilate most of the RBCOD in the anaerobic step. Thus a lower SDNR would be observed in the anoxic zone, due to less substrate availability and less denitrifying bacteria.

The second way BPR can affect the SDNRs in anoxic zones is if PAOs can use nitrate as an electron acceptor. The PAOs as called denitrifying PAOs (DPAOs). The DPAO uptake RBCOD in the anaerobic zone and convert it to PHB. The process of converting

and storing the RBCOD as PHB is believed to be less efficient than direct RBCOD uptake and utilization by non-PAO heterotrophs in the anoxic zone. PHB oxidation in the anoxic zone should result in a lower SDNR.

Research by Murakami et al. (1998) used bench-scale activated sludge units using anaerobic-anoxic selectors to examine the effects of changing selectors on both phosphorus and nitrogen removal. Results indicated that both nitrogen and phosphorus removal can be achieved with anaerobic-anoxic selectors. However, when the selectors were sized to favor biological phosphorus removal, nitrogen removal was reduced and vice versa.

Nicholls et al. (1983, 1987) noted that BPR-BNR systems produced higher denitrification rates than BNR systems only regardless of the whether or not nitrate was introduced to the anaerobic zone. This was not expected because the PAOs were assumed to utilize all the RBCOD in the influent wastewater leaving only SBCOD for denitrifying bacteria. Without the RBCOD, the denitrifiers were expected to grow at a slower rate resulting in lower SDNRs. Clayton et al. (1991) also observed that BPR-BNR systems stimulated higher denitrification rates than those reported in regular BNR systems. Clayton found that in a bench scale staged reactor system with anaerobic, anoxic, aerobic, and anoxic zones (BPR modified Bardenpho process) the RBCOD concentration driven denitrification rates were the same for a BNR and BPR-BNR system. However, SBCOD utilization in the primary and secondary anoxic tanks was almost 2.5 times higher than SBCOD utilization within BNR systems without BPR. The increased usage of SBCOD was assumed to more than offset the RBCOD utilization reduction due to PAOs RBCOD uptake in the anaerobic zone. The exact cause of the increased SBCOD utilization was not determined.

Other recent research also indicates that BPR-BNR systems achieve higher SDNRs. Research by Stevens et al. (1997), Filipe et al. (1997), Kerrn-Jespersen and Henze (1993), Bortone et al. (1996) suggests that two different populations of phosphorus accumulating

organisms (PAO) exist in BEPR systems. These two populations are believed to consist of PAOs that can use either nitrate or oxygen as electron acceptors or PAOs that can only use oxygen. Stevens et al. (1997) determined that denitrifiers were likely responsible for biological phosphorus removal at the Westbank, British Columbia plant that uses a 3-stage Bardenpho system. Stevens also noted that denitrification rates increased when primary effluent was added directly to the anoxic zones.

Filipe et al. (1997) modeled a denitrifying PAO and aerobic population in a sequence batch reactor system with anaerobic, anoxic and aerobic steps to explore the effect of population dynamics on nitrogen and phosphorus removal efficiency. A basic concept in the model was that aerobic oxidation of stored PHB by the PAOs results in a higher cell yield due to the larger change in free energy for oxygen versus nitrate as an electron acceptor. This assumption predicts that the aerobic only PAOs will eventually dominate the BPR system and denitrification rates in the anoxic zone would then be much lower than for a system without the first anaerobic zone to promote BPR. For their model simulations, the only time the facultative PAOs could dominate was when the aerobic HRT was relatively low (0.5 hours) and the system SRT was low (5 days). With longer SRTs, as well as longer aerobic periods, the aerobic-only PAOs would out compete the facultative PAOs. These results do not explain why full-scale BPR plants achieve high SDNRs and maintain facultative PAOs as indicated by phosphorus uptake concurrent with nitrate removal in the anoxic zones (Stevens et al. (1997); Reardon et al. (1996); Rabinowitz et al. (1986)).

2.11 IAWQ General Model for Biological Nutrient Removal in Activated Sludge Systems

The BioWin model as described by Barker and Dold (1997a,b) conforms to the IAWQ activated sludge modeling recommendations and can be used to predict the biomass populations and performance in BNR and BPR systems. This model is based upon the General Model for Biological Nutrient Removal in Activated Sludge Systems developed

through the efforts of the International Association on Water Pollution Research and Control (IAWPRC, now IAWQ) and a modified version of the Wentzel et al. (1989a,b) model for biological phosphorus removal. The preliminary version of the IAWPRC model was developed and presented by Dold and Marias (1986) for a comprehensive evaluation by the IAWPRC. The final version of the model is called the IAWPRC Model No 1 (ASM1) in 1987. This model is based on a mechanistic description of organism growth rates and substrate stoichimetry to predict responses for a wide ranges of activate sludge system configurations, influent wastewater characteristics, and operational parameters (Barker and Dold, 1997a,b). The model has equations for carbonaceous removal, nitrification, denitrification, and phosphorus removing biological processes. System configurations can include single and series reactors, aerated and non-aerated reactors, and inter-reactor recycles. Influent wastewater characteristics included RBCOD and SBCOD, TKN, NH_4-N, phosphorus, inert VSS, and variable influent flow patterns. Important operational parameters are SRT, temperature, and DO concentration. Influent COD substrates are divided into four fractions in the model: short chain fatty acid COD (S_{BSA}), readily biodegradable 'complex' COD (S_{BSC}), slowly biodegradable (enmeshed) COD (S_{ENM}), and slowly biodegradable COD (S_{BS}). The readily biodegradable fraction is hypothesized to consist of material that can be absorbed readily by the organism and metabolized for energy and synthesis. The slowly biodegradable fraction is assumed to be made up of particulate/colloidal material and complex organic molecules that require extracellular enzymatic breakdown prior to adsorption and utilization. Growth of the non-PAO heterotrophs on 'complex' readily biodegradable COD is modeled by the Monod relationship. The S_{ENM} is assumed to be enmeshed in the sludge mass. The S_{ENM} is broken down extracellularly and added to the wastewater's pool of S_{BSC}. The rate of hydrolysis of the S_{BSC} under anaerobic conditions is considered to be a fraction of the rate under aerobic conditions ($\eta_{S,ANOX}$). The biodegradation of the S_{BS} to S_{BSC} is assumed to occur at the same rate as that for the S_{ENM}. The model incorporates assumptions for biological reaction coefficients, and many of these are related to denitrification rates, the subject of this research. The relevant parameters in the model that are important for the

performance of anoxic reactors are listed below with their default values (Barker and Dold , 1997a,b).

Table 2.9 BioWin Model Parameters Important for Anoxic Zone Reactions with Default Values

Parameter	*Description*	*Value*	*Units*
$Y_{H,ANOX}$	Yield (anoxic)	0.666	g cell COD yield/g COD utilized
E_{ANOX}	Hydrolysis efficiency (anoxic)	0.90	g S_{BSC} / g S_{ENM}
$F_{N,ZH}$	Nitrogen content of active mass	0.068	g N/g COD active organisms
$F_{N,ZEH}$	Nitrogen content of endogenous mass	0.068	g N/g COD endogenous residue
$F_{EP,H}$	Frac of active mass remaining as endog	0.08	g COD endog mass/g COD active mass
$F_{CV,H}$	Ratio COD/VSS	1.48	g COD/g VSS
Hydrolysis Rate	Hydrolysis Rate of SBCOD to	2.81	g COD/g VSS
K_s for hydrolysis	Half velocity constant for hydrolysis	0.15	mg/L
	Neta anoxic hydrolysis	1.0	
Mu Max	Maximum specific substrate utilization (RBCOD)	3.2	mg/L
K_s	Half velocity constant for RBCOD utilization	5.0	mg/L
η_g	Heterotrophic fraction denitrifying	0.37	g denitrifying biomass. g non-PAO heterotrophs

Where: S_{BSC} = readily biodegradable 'complex' COD

S_{ENM} = slowly biodegradable enmeshed COD

One of the key features of this model is that it recognizes that the active biomass is only a portion of the activated sludge MLVSS concentration and calculates the active VSS concentration based on the COD removed, SRT, endogenous decay coefficient, yields, etc. It accounts for non-degradable VSS in the wastewater to determine the inert fraction of the activated sludge MLVSS concentration. Whether or not a treatment plant has primary treatment impacts the active VSS fraction since primary treatment removes a large portion of inert VSS from the wastewater. The composition of the active VSS is

divided into three different bio-populations: nitrifiers, heterotrophic PAOs, and heterotrophic non-PAOs. The model further divides the total heterotrophic biomass into fractions that can and cannot use nitrate as an electron acceptor. This fraction greatly affects anoxic zone SDNRs and at present the model is not able to predict this fraction. A default value of 0.37 (η_g) is presently assumed meaning that 37 percent of the heterotrophic biomass is able to denitrify.

The model assumes two yield coefficients for the non-polyP heterotrophs to account for the possibility of lower yields under anoxic condition versus aerobic conditions. These yields are $Y_{H,AER}$ and $Y_{H,ANOX}$.

In the absence of DO, only a portion of the non-polyP heterotrophic population is capable of using nitrate, if available. So under anoxic conditions the substrate utilization rates are adjusted by factor (η_g) to account for reduced rate of substrate removal under anoxic conditions. Decay of the heterotrophs is modeled according to the death-regeneration theory. According to this theory the bacteria are assumed to die at a rate that is proportional to the active biomass concentration. During cell death a portion of the biomass COD is assumed to be released as particulate biodegradable COD (S_{ENM}).

As discussed earlier, many factors have been reported to inhibit the denitrification process. The ASM2 model has incorporated so called switching functions to ensure inhibition is accounted for within the model. Switching functions are used as mathematical switches for turning processes on and off when concentrations of selected components are above (switch on) and below (switch off) the threshold magnitudes. Within anoxic zones a switching function is used for DO to indicate when this parameter can impair denitrification. The switching function is given in Equation 2.9.

$$q_D = q'_D \times \left(\frac{K_o}{K_o + S_o} \right) \qquad (2.9)$$

Where: q_D = nitrate removal rate, g NO3-N/g VSS-d

q'_D = maximum specific nitrate removal rate, g NO3-N/g VSS-d

K_o = half-saturation constant for oxygen in mg/L

S_o = DO concentration in mg/L

As seen from Equation 2.8 zero DO concentrations have no affect on nitrate removal rates. As DO concentrations increase, the nitrate removal rate decreases. K_o is set at 0.05 in the ASM2 model equating to a 50% decrease in denitrification rates at DO concentrations of 0.05 mg/L.

The BioWin model provides an accepted mechanistic model for activated sludge system designs that can include an anoxic zone design. Many assumptions are necessary in the model and the model authors recommend an extensive calibration procedure to better quantify important model coefficients for different locations and process configurations.

2.12 Conclusions

The literature has shown two approaches that have been used to size anoxic basin reactors in BNR activated sludge systems. One of these involves a simple calculation based on using SDNRs obtained from bench-scale or full-scale test results and the other involves the use of a mechanistic activated sludge model (IAWQ model) that accounts for the wastewater characteristics, biological kinetics, and biological population composition. Reported SDNR values within the literature vary widely and the influence of wastewater characteristics, temperature, and parameters that effect the active denitrifying biomass in the MLVSS on observed SDNRs is not clearly understood. The database for substrate utilization kinetics in the IAWQ model is limited, and site specific model calibration efforts are recommended. Key parameters that appear to be critical to anoxic zone denitrification rates in all cases are the wastewater COD composition in terms of RBCOD

and SBCOD, the active biomass fraction in the activated sludge mixed liquor capable of denitrification, the anoxic zone HRT, temperature, wastewater source, and substrate utilization bio-kinetic coefficients for anoxic reactions. The research for this thesis was intended to provide information on the effect of wastewater characteristics and anoxic zone HRT on observed denitrification kinetics and SDNRs.

Chapter 3: Experimental Description

3.1 General Approach

The objective of the research was to observe denitrification kinetics for different municipal wastewaters under conditions representative of full-scale plant anoxic tank design hydraulic detention times (HRTs) and to relate the kinetics to fundamental process parameters. Fundamental process parameters of interest were the RBCOD and the total biodegradable soluble COD (TBSCOD) of the wastewater, and the active biomass fraction capable of nitrate reduction for mixed liquor developed from the full-scale activated sludge treatment systems. One of the criteria for selecting the municipal plants for this study was that the plant had biological nitrification and denitrification occurring.

A continuous flow, completely mixed test reactor was operated under anoxic conditions at four municipal wastewater treatment plants (WWTPs) to observe denitrification rates. The reactor received influent wastewater, return activated sludge (RAS), and nitrate. The feed flow rate was used to control the anoxic reactor hydraulic detention time (HRT). Specific denitrification rates (SDNRs) were determined as a function of HRT at steady state conditions. The reactor was operated at constant flow and constant influent wastewater and RAS feed conditions for a time equal to three or more times the HRT to achieve steady state prior to data collection. At least three different HRT levels were selected for each site to obtain a range of SDNRs. The SCOD, TBSCOD, and RBCOD concentrations were determined for the wastewater feed within the reactor at steady state conditions to develop a relationship between these concentrations and the observed denitrification rates. For each site the active biomass fraction of the mixed liquor was estimated from the site information provided on the wastewater characteristics, activated sludge system HRT and SRT, average amount of solids wasted per day, and mixed liquor TSS and VSS concentrations.

Laboratory batch tests were carried out to determine the active denitrifying biomass fraction in the mixed liquor for each WWTP. Batch tests were conducted with the activated sludge from each site. The active fraction of denitrifying biomass was based on the ratio of the observed equivalent oxygen uptake when using nitrate for the electron acceptor to that for oxygen during endogenous respiration.

The on-site continuous flow reactor method used had advantages over the batch denitrification tests used by previous investigators (Ekama et al., 1986). With the reactor test technique used, COD and nitrate mass balances and substrate measurements not possible for full-scale plant testing could be done. This also overcomes one of the problems with batch testing for denitrification rates that start with high initial substrate concentration, which is not representative of conditions that actually occur in anoxic reactors. Continuous flow reactor tests provide results from operations that mimic possible full-scale plant anoxic reactor operating HRTs and thus practical site kinetic conditions.

Full-scale plant data was also obtained at one of the plants to compare COD utilization rates to observed SDNRs.

The experimental plan describes the field testing reactor and its operation and sampling and analytical program, the laboratory bench scale test procedure to observe endogenous oxygen uptake and denitrification rates, full-scale plant sampling and analyses, preparation of samples for analytical analysis, and analytical methods used in this research.

3.2 Wastewater Plants Used for Site Testing

Anoxic reactor experiments were carried out at four different WWTPs to determine if different wastewater sources would affect denitrification kinetics. The activated sludge of these plants was used in the anoxic reactor. Thus one plant selection requirement was

having denitrification activity within their activated sludge system so that a biological population capable active denitrification was assured for the test work. The return activated sludge (RAS) from the secondary clarifiers was used to feed the anoxic test reactor along with the influent wastewater at the plant. The plants selected are listed in Table 3.1.

Table 3.1 Plants Used to Provide both RAS and Wastewater for Anoxic Reactor Testing

Plant Name	*Location*	*Treatment Process*
Olympus Terrace	Muckleteo, WA	On-off Aeration Oxidation Ditch with Anoxic Selector
Snoqualmie Falls	Snoqualmie Falls, WA	Anaerobic-Anoxic-Aerobic Treatment
Chambers Creek	Tacoma/Steilacoom WA	Staged Anoxic-Aerobic
LOTT	Olympia, WA	Stage Anoxic-Aerobic-Staged Anoxic-Aerobic Process

Both the Chamber's Creek and LOTT plants have staged A/O activated sludge processes. Both plants also have primary clarification. The Chamber's Creek plant uses a four stage anoxic system, followed by a primary aeration tank and polishing aeration tank. The system SRT averages about four days in the summer. The LOTT has a Bardenpho process consisting of four equally sized stages in the first anoxic zone, a five-stage primary aeration tank, and a four stage anoxic tank second anoxic zone, followed by a short aeration zone before secondary clarification. The plant operates at an average SRT of approximately 12 days during the summer months.

The Olympus Terrace and Snoqualmie Falls plants are oxidation ditch systems. Anoxic conditions are developed at the Olympus Terrace plant by turning the aeration on and off during the day. A submerged mixer maintains the activated sludge and wastewater (mixed liquor) in suspension when the aerators are turned off and thus the tank operates as an anoxic reactor under those conditions. An ORP probe measurement determines when the nitrate is consumed and on that basis the aeration is then resumed. Normally, one 'aeration-off' cycle is used per day lasting from 5 to 6 hours in duration. The average

operating SRT of this plant is 13 days during summer operations. The Snoqualmie Falls plant had an anaerobic contact time of 8.2 hours and an anoxic zone contact time of 21 hours ahead of the oxidation ditch based on the plant's August 1998 flows. Mixed liquor from the oxidation ditch is recycled to the anoxic zone. Nitrate reduction occurs in both anoxic zone and in the ditch mixed liquor downstream from the aeration zone. This plant is currently under loaded (10% of design flows) and uses an operating SRT 60 days. During the anoxic reactor experiments conducted the plants SRT ranged from 20 to 30 days.

A complete description of the test site facilities is contained in Appendix 5 and describes the plant flow rates, discharge permit requirements, treatment performance, and influent wastewater characteristics.

3.3 Field Test Reactor Description

A test reactor was designed and operated to obtain denitrification rates at operating HRTs possible for full-scale WWTP anoxic reactors.

The reactor system used for field site testing in this research is shown in Figure 3.1:

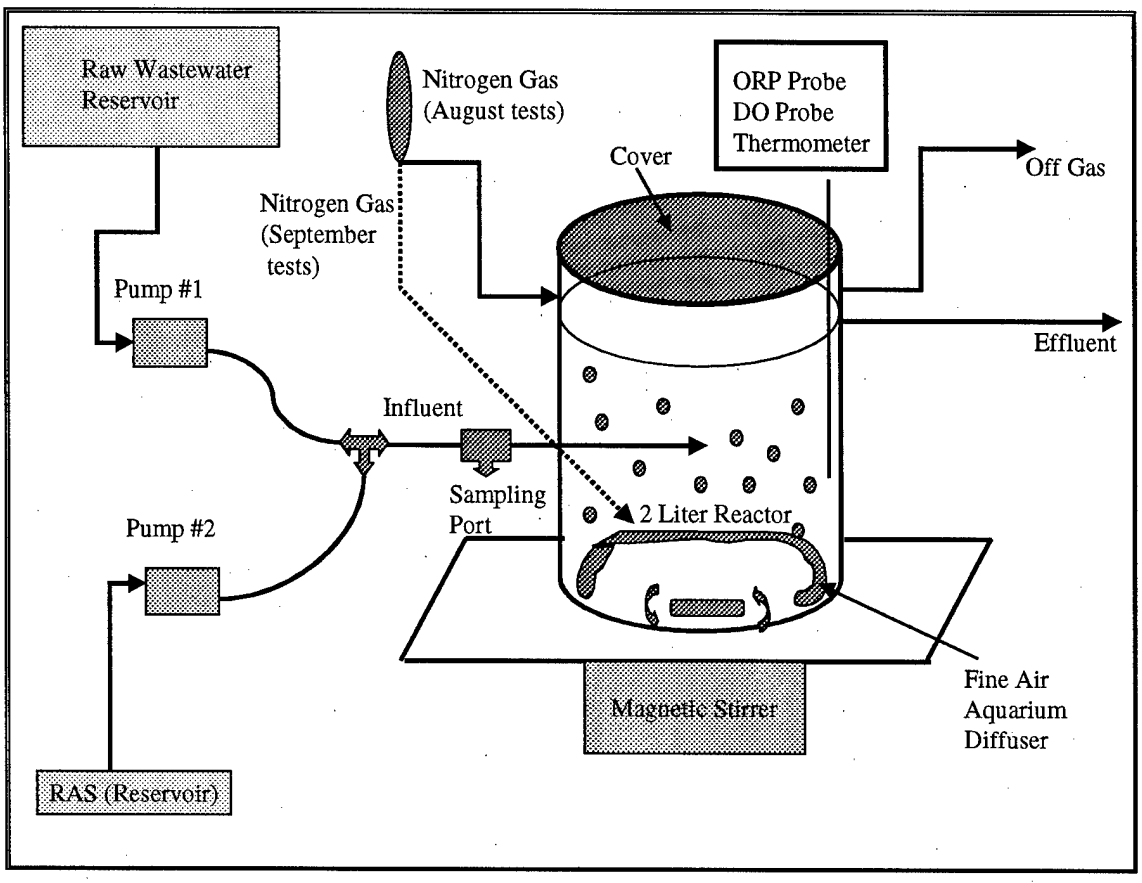

FIGURE 3.1 Anoxic Reactor Used for Site Denitrification Rate Testing

The reactor was 30 cm (12 inch) in diameter by 20-cm (8 inch) tall and was made of a 32-mm (1/8-inch) thick Plexiglas cylinder with 32-mm thick Plexiglas bottom and top plates. The total volume was approximately 2.7 L, and a 64-mm (¼ inch) effluent hole was located to provide an operating volume of 2 liters. The top of the reactor was covered to prevent oxygen introduction. Half of the top was permanently sealed and the other half of the cover's lid was on a hinge to allow access for sampling and cleaning. Nitrogen gas was sparged at approximately 2 liter per minute into the liquid's surface during the July and August testing and then fed through the headspace by an aquarium membrane fine air diffuser during the September tests at one liter per minute. A 60-CF industrial nitrogen gas cylinder was used as the nitrogen gas source. Gas escaped from the reactor through a 64-mm hole drilled in the reactor's lid and through the effluent line. A 1.25-cm (½ inch) hole drilled through the reactor lid allowed insertion of ORP and pH probes for on-line

monitoring. Periodically, the probe was removed and a thermometer mounted in a rubber stopper was inserted for temperature readings. A 1.24-cm influent tap was located one inch below the 2-liter water level of the reactor. Both the influent and effluent orifices were fitted with 1.24-cm plastic irrigation line fittings. Clear rubber tubing was connected to the effluent fitting to allow the effluent to gravity flow the effluent receptacle.

A 75.7-liter (20-gallon) plastic trash bucket was used as the wastewater feed reservoir. Raw wastewater, sodium nitrate, and acetate (when added) were mixed with a plastic and PVC pipe in the wastewater feed reservoir and was fed to the reactor by a master flex peristaltic pump through 32-mm to 48-mm (3/16 inch) tygon tubing. The pump rpm and tubing diameter were used to control the flow rates from the wastewater reservoir to the reactor. No mixing occurred in the wastewater feed reservoir once the feed source was prepared. An 18.9-liter (five-gallon) plastic bucket was used for the WWTP return activated sludge (RAS) reservoir. The RAS reservoir was continually mixed with a 1/20 HP electric mixer using a 60-cm aluminum rod with 5-cm flat propeller. A master flex peristaltic pump fed the RAS to the reactor through 32-mm to 48-mm tygon tubing. The pump rpm and tubing diameter used for the RAS container controlled the flow rates, and thus a portion of the liquid detention time within the anoxic reactor, from the wastewater reservoir to the reactor. Both the RAS and wastewater feed lines joined into one 30-mm feed line prior to entering the anoxic reactor. A 32-mm irrigation tee fitting was used to join the feed lines to the influent line. A 64-mm in line sampling tee was inserted in the influent line between the tee and reactor. The reactor was mixed with a magnetic stirrer using a 2.5-cm (½ inch) TEFLON stirring bar.

3.3.1 Field Reactor Feed Sources, Reactor Preparation and Flow Rates

Raw wastewater was obtained from the head works of each WWTP and prior to any primary treatment for plants with primary treatment (LOTT and Chamber's Creek). Effluent samples used for non-biodegradable COD (NBSCOD) quantification from each

of the WWTPs were taken just prior to the disinfection step except for Snoqualmie Falls. Snoqualmie Fall's effluent samples were taken after sand filtration, and the COD concentrations for soluble and flocculated CODs were too low to be determined. The results indicated the sand filter captured the soluble NBSCOD remaining within the effluent after clarification. Accurate NBSCOD quantification was required for determining the actual biodegradable COD available for cell growth. Therefore, Snoqualmie Fall's effluent samples were taken directly from the clarifier effluent.

Each plant reported a slight variation in wastewater and activated sludge characteristics from the weekdays to the weekend. To reduce the effects of this variance in characteristics on the experimental results, anoxic field reactor testing was generally conducted at the WWTPs from Tuesday to Thursday.

Wastewater was collected in five-gallon buckets and then added to the wastewater feed reservoir until a volume of 44 to 48 liters was obtained. The reservoir volume was measured using a hollow 1.27-cm (½ inch) PVC pipe that was marked in 2-liter increments. The PVC measuring pipe was calibrated by recording the water level during each addition of water from a 2-liter volumetric flask. Sodium acetate was added to the wastewater container to obtain a 20 mg/L concentration. Nitrate was added as sodium nitrate at an initial concentration of 38 mg/L. After nitrate or acetate addition, the wastewater was mixed with PVC tubing for approximately one minute. No mixing was done during the experiments. The feed was taken at approximately 20 cm above the bottom the reservoir so that no appreciable settled solids were taken.

Activated sludge was obtained from the recycled activated sludge (RAS) lines for each plant except Snoqualmie Falls. The low RAS concentration in the first reactor tests at the Snoqualmie Falls (SF) WWTP resulted in very small amounts of nitrate removal in the anoxic reactor. To increase the denitrification activity within the anoxic reactor, the RAS

from SF WWTP was further concentrated for future tests by settling for a few minutes and decanting supernatant.

RAS was collected in five-gallon buckets from the WWTP return sludge line and then added to the RAS container. Liquid displacement was measured with the calibrated hollow PVC pipes. The RAS measuring stick was calibrated to the nearest liter and measured to the nearest 1/10 of a liter. Sodium nitrate was added the RAS reservoir at an initial concentration of about 10 mg/L to assure ensure the nitrate reduction activity was maintained within the RAS. The RAS was continuously mixed with a 1/20 horsepower electric mixer. The RAS reservoir did not appear to be oxygenated due to mixing based on no visual vortexing from the mixing, a foam cover that quickly developed at the of the reservoir and negative ORP measurements taken within the reservoir.

Different anoxic reactor HRTs were used to vary the steady state substrate loading conditions. The minimum HRT used was 10 minutes. The HRTs initially chosen were 10, 15, 20 and 30 minutes. A total influent feed flow rate was determined for each HRT based on the reactor 2-liter volume. The target feed flow rates consisted of about 70% wastewater and 30% RAS. Based on the desired feed flow rates, the pump controller speed was estimated and set.

Actual feed flow rates were measured by two methods. The first method measured the total reactor outflow. This consisted of measuring the time required for the effluent to fill a 250-mL volumetric flask. The outflow was assumed to be equal to the influent total flow rate. Individual wastewater and RAS flow rates were determined by recording the liquid volume changes in each of the feed reservoirs over time. The sum of the RAS and wastewater feed flow rates based on reservoir volume changes were compared to the effluent flow rates manually measured. A good balance (within 5%) was always found.

The reactor was expected to reach steady state conditions by an operating time equal to 3 HRTs. During the first set of experiments, samples were taken after either 3 and 3.5 HRTs of time passed or after 3.5 and 4.0 HRTs of time passed. Samples taken after 3.0 and 3.5 HRTs showed some variation in COD and nitrate concentrations. Samples taken after 3.5 and 4.0 HRTs time periods were virtually identical when analyzed. Therefore, a 3.5 HRT time period was chosen in subsequent experiments to achieve steady state conditions. ORP activity was also closely monitored over the last HRT time period and were found to stabilized by then. Typical ORP values were –29 to –40 mV.

3.3.2 Sampling Frequency and Handling Methods

The sampling frequency and analysis used for each sampling location are summarized in Table 3.2.

Table 3.2 Sampling Plan for Field Reactor Denitrification Tests

Sample Analysis	Sample Location				
	Wastewater Feed Reservoir	RAS Reservoir	Influent Sample	Reactor	Effluent
Sample Type:					
ORP	SS(1)	SS(1)		D/SS(1)	
pH	SS(1)	SS(1)		D/SS(1)	
Temperature	SS(1)	SS(1)		SS(1)	
Flow Rates	SS(1)	SS(1)			I/SS(1)
Analyses Conducted:					
TSS/VSS		I(2)*		SS(2)**	
TCOD	I(1)*		SS(1)**		
SCOD	I(1)*	I(1)*	SS(1)**		SS(1)
FCOD***	I(1)*		SS(1)**		SS(1)
NO_3			SS(2)		SS(2)
NO_2			SS(1)		SS(1)
Alkalinity			SS(1)		SS(1)
PO_4			SS(1)		SS(1)

Symbols used in Table 3.2:
I – Sample taken at start of experimental run, number of samples in parenthesis.
SS – Sample taken at 'steady state' conditions, number of samples in parenthesis.
D – Measurements monitored during experimental runs.
* - Indicates location of samples taken during August site experiments only
** - Indicates location of samples taken during September site experiments only
***-Flocculated COD (FCOD) determined by adding zinc sulfate to sample and raising the sample pH to 10.5 to cause precipitation of colloidal and other suspended solids prior to 0.45μ membrane filtration and COD analysis. This is discussed in detail later.

ORP, temperature, and pH were measured for each steady state condition. For all steady state sampling the reactor effluent samples were collected first followed by reactor sampling (September only), and ending with the influent sample collection. Influent samples were taken after reactor and effluent to ensure the reactor steady state conditions were not disrupted prior to effluent and reactor sampling.

Reactor samples were taken with 10-mL syringes. The differences in COD and VSS sampling location between August and September were due to a change in sampling protocol. The protocol was changed to get more direct and accurate COD and VSS measurements. In July, COD measurements were taken from the wastewater and RAS reservoirs and the influent COD was calculated based on the concentrations and reservoir feed rates. The MLVSS sample was taken in the RAS container and the reactor MLVSS was estimated based on the reactor feed rates assuming the wastewater reservoir VSS was zero. Influent samples were taken from the tee fitting directly after the RAS and wastewater feed flow joined and before entering the reactor. Effluent samples from the reactor were taken from the reactor effluent line. All influent and effluent reactor samples except for the TCOD and MLVSS samples were immediately pre-filtered by passing the sample through a coffee filter in a plastic funnel to quickly remove most of the biomass and other suspended solids. Each influent and effluent sample was 250 mL.

The September experiments gathered more data on denitrification kinetics while improving the quantification of the MLVSS in the reactor, and the SCOD and RBCOD in the reactor influent. MLVSS samples (5 to 10 mL) were taken directly from reactor after steady conditions were achieved. COD measurements were taken from the influent samples taken at the influent tee. These samples provided a more direct quantification of MLVSS within the reactor and CODs in the influent for each HRT. The same sample size and preparation used for the August samples applied to the September samples.

All the MLVSS samples were prepared on site. The MLVSS samples were filtered at the WWTPs using pre-weighed filters and the plant's vacuum/filtering apparatus. The Gelman Sciences Type A/E 47-mm glass fiber filter papers were prepared at the university laboratory prior to the onsite experiments. The filter paper was prepared by rinsing the paper with approximately 100 mL of deionized water, drying in a 105° C oven for at least 30 minutes, igniting at 550° C for at least 20 minutes, and then cooling in a dessicator for more than 30 minutes. The initial weight of the filter was measured in a tin and recorded.

Solids quantification (to include drying) occurred at the university laboratory. To determine the TSS for a sample, a measured volume of sample was then filtered through the prepared and weighed filter. A graduated 10 mL syringe was used to measure the volume of sample, and the syringe was refilled with deionized water and to ensure all of the biomass in the sample was passed out of the syringe and collected on the filter paper.

For each experimental run in August and September, two effluent and two influent samples were taken. All samples except SCOD were immediately pre-filtered by passing the sample through a coffee filter in a plastic funnel to quickly remove most of the biomass and other suspended solids. The influent and effluent samples were split and handled differently depending on the analyses to be conducted. The pH of one of the split samples was reduced to below 2.0 by the addition of 0.1 mL of H_2SO_4. All samples were then put on ice to reduce sample temperature until the samples could be stored in the laboratory refrigeration unit the next day at 4°C. The low pH sample was used for nitrate and COD analyses. The other sample was used for nitrite, alkalinity, and phosphorus analyses.

All samples except for the SCOD were prepared prior to analysis in the laboratory by warming to room temperature and filtering through a 0.45 um Millipore filter paper. Samples with reduced pH were raised to neutral pH by adding 5 M NaOH drop-wise until a pH of 6 to 6.5 was obtained prior to filtration. Nitrite samples and ortho-phorphorous samples were diluted with deionized water at a ratio of 1:20.

The flocculated soluble COD test was used to measure the RBCOD concentrations in the influent and effluent and used to determine denitrification substrate utilization rates. Samples were prepared using the "Rapid Physical-Chemical Method – The Flocculation

Method" (RPCM-F method) Mamais et al. (1993). The reactor digestion method, using the HACH Low Range (0 to 150 mg/L COD) reagent vials, was used for actual measurements.

The non-degradable portion of soluble COD is determined by conducting the flocculation test on a WWTP effluent sample from a clarifier. By subtracting the results of flocculated effluent sample considered to be the NDSCOD from the total 'truly' soluble COD the resulting RBCOD was determined.

Using the RPCM-F method, filtered samples were flocculated by adding 1 mL of a 100-g/L zinc sulfate solution to a 100 mL wastewater sample in 125 mL beaker, followed by at least two minutes of vigorous mixing. The sample's pH level was adjusted to approximately 10.5 using a 6-M sodium hydroxide solution drop wise. Real time pH was monitored using the Beckman 200 pH meter and Corning electrode. A few minutes were allowed for quiescent settling of the flocculated sample. Next, 20-25 mL of clear supernatant was drawn into a graduated syringe and then passed through a 0.45 um Millipore filter paper to remove colloidal particles captured by precipitated $Zn(OH)_{2(s)}$ (Mamias, 196). The sample was then analyzed for COD concentration.

The plant's effluent FCOD was determined in the same way outlined above. Mamias et al (1993) reported that the plant effluent for WWTPs with at least a five-day SRT provided a reasonable estimate of the NBSCOD concentration for the wastewater. The sample RBCOD was then determined as the sample FCOD minus the plant effluent's FCOD as outlined in Equation 3.1:

$$RBCOD_{sample} = FCOD_{sample} - FCOD_{Plant\ Effluent} \qquad (3.1)$$

3.4 Laboratory Tests to Determine the Active Biomass

Only a fraction of the bacteria comprising the biomass is generally considered capable of denitrifying. To accurately reflect the biomass responsible for denitrification, the denitrifying fraction must be determined. This correction is assumed to be 37 % in the

BioWin Model. However, based upon wastewater characteristics and WWTP operating parameters, the denitrifying fraction may vary between WWTPs. This section outlines how waste activated sludge sample from the different WWTPs were analyzed in the university laboratory reactor to estimate the denitrifying fraction of the heterotrophic bacteria within the mixed liquor.

An estimation of the fraction was made by determining the ratio of the mixed liquor's oxygen equivalent specific endogenous nitrate utilization rate ($SDNR_{endog}$) to its specific endogenous oxygen utilization rate (SEOUR). The $SDNR_{endog}$ is the rate of nitrate utilization during endogenous respiration in the absence of DO. The $SDNR_{endog}$ /SEOUR fraction was corrected by 2.86 g O_2 / g NO_3 for oxygen equivalency to yield the oxygen equivalent consumption ratio.

$$\text{Equivalent } O_2 \text{ Use Ratio} = 2.86\ SDNR_{endog} / SOUR \quad (3.2)$$

The fraction of the test reactor mixed liquor capable of using nitrate for respiration in the absence of oxygen was assumed to be proportional to the equivalent oxygen use ratio.

$$\frac{(SDNR_{endog} \text{ in g } NO_3/\text{g VSS-d})*2.86}{(\text{SEOUR in g } O_2 / \text{g VSS-d})} \qquad (3.3)$$

To calculate the equivalent O_2 use ratio and the fraction of denitrifying bacteria, both the $SDNR_{endog}$ and SEOUR must be determined. The following subsections describe how experiments were conducted to determine both the $SDNR_{endog}$ and SEOUR.

The SEOUR was determined for each WWTP's mixed liquor. A one liter mixed liquor sample from each WWTP was used in this experiment. The mixed liquor was obtained from the WWTP RAS line during the anoxic reactor field experiments conducted in August. Samples were collected and stored in a one-liter dark brown glass container. The

sample was stored in an ice chest until transferred to 4 °C cold storage at the university lab. The actual SEOUR testing was conducted the next day.

When the OUR test was conducted, the sample was pre-aerated for at least four hours during which it warmed to room temperature. After the four hours of aeration, approximately 10 to 15 mg/L of nitrate was added in the form of sodium nitrate to the mixed liquor. After an additional hour of aeration the OUR test was conducted. The long storage and subsequent aeration period was assumed adequate to allow the mixed liquor ample time to degrade any degradable soluble COD produced within the mixed liquor during storage.

During the OUR test, a 250-mL BOD bottle was filled with mixed liquor. A TEFLON stirring rod was inserted into the container and the container was capped with a DO probe. A DO meter was used to measure the DO concentration within the BOD bottle over time. The DO concentration over time was recorded once a minute until the DO reached less than 1.0 mg/l. The SEOUR was calculated as the specific oxygen consumption rate as shown in Equation 3.4 below:

$$\text{SEOUR} = \frac{(\Delta \text{ oxygen in mg/L})*(60 \text{ min/hr})*(24 \text{ hr/day})}{(\text{Length of test in min})*(\text{VSS concentration in mg/L})} \qquad (3.4)$$

The mixed liquor's VSS was determined for two samples taken from the OUR test bottle at the end of the test. Equation 2.3 and the standard temperature correction coefficient used in the BioWin model (θ =1.029) were used to correct the SEOUR rate to 20 °C.

The NUR was determined using an endogenous denitrification test reactor as shown in Figure 3.2.

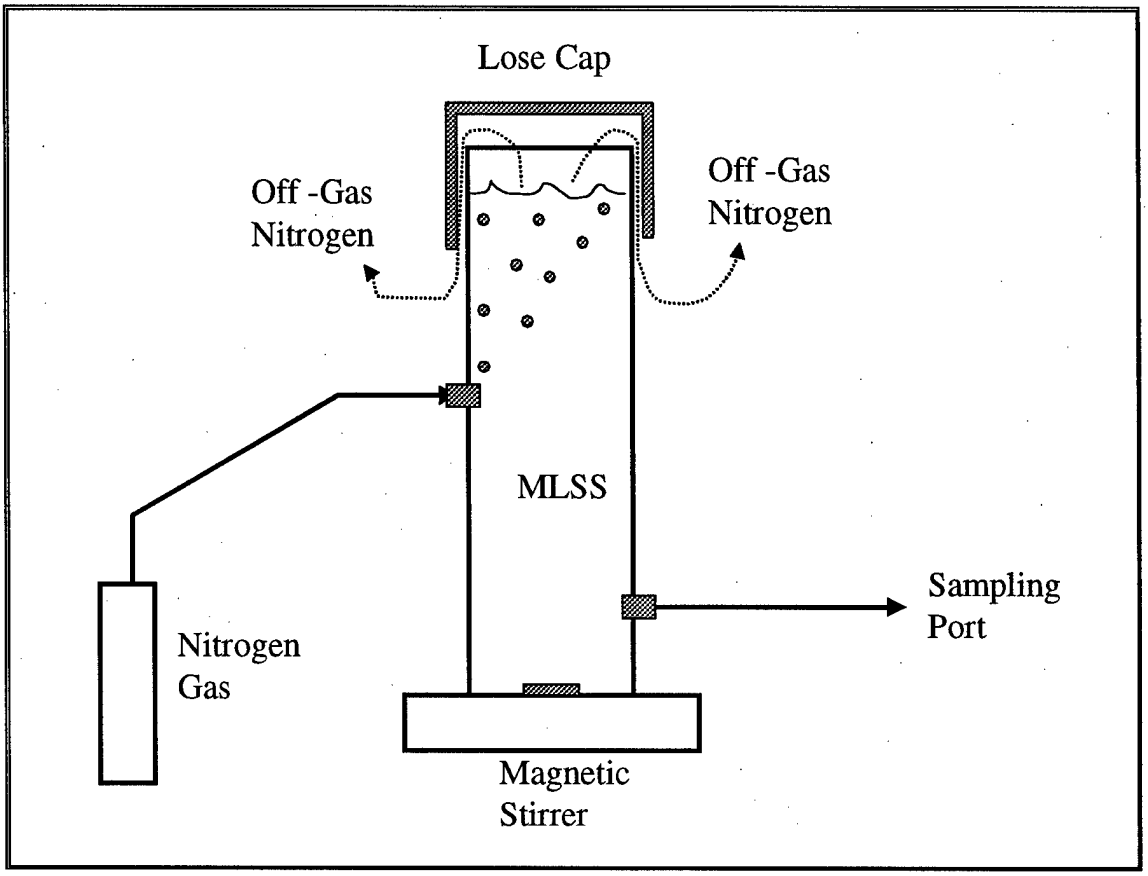

Figure 3.2 Endogenous Respiration Reactor

The reactor consisted of a Plexiglas cylinder attached to a 1.27-mm (½ inch) thick Plexiglas block with caulking. The reactor was 10.16 cm (4 inches) in diameter and 25.4-cm tall with an internal liquid volume capacity of 500-mL. A magnetic stirrer was used

to maintain the mixed liquor in suspension. After the mixed liquor sample was added to the reactor, a loose fitting cap was placed over the reactor. The top half of the mixed liquor was sparged with nitrogen gas to prevent aeration of the mixed liquor at the surface of the reactor. Samples were taken from the sampling port located three inched from the bottom of the reactor.

The same mixed liquor used in the SEOUR determination test was used in the NUR test. Since the endogenous respiration reactor had a volume of 500-mL, additional mixed liquor

was needed to conduct the NUR test. An additional 250-mL of sample was prepared in the same way as the first sample and was aerated for the same time and mixed during the OUR test. After the OUR test was completed, an additional 30 minutes was allowed for the DO to reach 0.0 mg/l. After this thirty-minute period, the 250-mL SOUR test sample and the additional mixed liquor sample were added to the endogenous anoxic reactor and the nitrate reduction rate test began.

At 20 to 25 minutes after the mixed liquor was added to the reactor, 15-mL samples were taken. Samples were pre-filtered with a coffee filter to quickly remove most of the solids. The coffee filtered effluent was then filtered with 0.45 μm Millipore filter paper. The sample was then analyzed for nitrate concentration. Samples were collected every 20 to 25 minutes and analyzed until the reactor's nitrate concentration dropped below 2 mg/L.

The change in nitrate over time divide by the MLVSS concentration is the $SDNR_{endog.}$

$$SDNR_{endog} = \frac{\text{(change in nitrate in mg/L)*(60 min/hr)*(24 hr/day)}}{\text{(Length of test in min)*(VSS concentration in mg/L)}} \quad (3.5)$$

The $SDNR_{endog}$ was also corrected to 20 °C using Equation 2.3.

3.5 Full-Scale WWTP Anoxic Zone COD Utilization and SDNRs

WWTP COD utilization tests were conducted at the Chamber's Creek WWTP. This treatment plant used a staged anoxic system that was well suited for monitoring actual plant substrate utilization rates. In this test the plants actual operating conditions such as flows, volumes, recycle rates, and RAS rates were obtained. The nominal liquid residence time (NRT) for each anoxic stage was determined. Samples were taken from successive treatment stages at times equal to the stage NRT after the previous stage sample. This method was expected to follow the mixed liquor flow through the anoxic stages.

Two mixed liquor samples taken in each treatment zone and were immediately pre-filtered with coffee filters to remove the majority of biomass and other suspended solids. The samples were then put in a cooler until analyzed later in the day. Concurrently, biomass samples were taken and filtered on site to determine the biomass present at each stage.

With the MLVSS, substrate, and nitrate concentrations measured, and residence time determined, the specific substrate and nitrate utilization rates within the treatment could be determined and compared with the results of the anoxic reactor site testing.

3.6 Analytical Methods

Table 3.3 summarizes the analytical methods used.

TABLE 3.3 Analytical Methods Used

Parameter	Method	Description
MLSS/MLVSS	Standard Methods	Methods 2540 D, 2540 E
COD	Hach COD reagents vials	Digestion Method 0-150/1500
Nitrite	Hach reagents	Diazotization Method
Nitrate	Hach reagents	Cadmium Reduction Method
PH/ORP	PH/ORP meter and probe	Direct Measurement
DO	DO Probe	Direct Measurement
Ortho-Phosphorus	HACH reagents	Ascorbic Acid Method
Alkalinity	Standard Methods	Titration Method 2320B

3.6.1 Biomass Measurement

Biomass was measured by dry weight. Both the total suspended solids (TSS) and volatile suspended solids (VSS) were measured. Weights were measured to 0.1-mg precision, and at all stages the filter papers and tins were handled only with tweezers.

3.6.1.1 Total Suspended Solids

The paper containing the retained matter prepared on site was dried at 105° C for more than 30 minutes, cooled in a dessicator for at least 30 minutes, and then weighed. TSS was the dry weight minus paperweight divided by the sample volume as shown in Equation 3.6 below.

$$\text{mg TSS/L} = \frac{((\text{weight of filter + dried weight}) - (\text{weight of filter})) \times 1000}{\text{sample volume in mL}} \quad (3.6)$$

3.6.1.2 Volatile Suspended Solids

VSS determination were performed on the TSS sample. After a sample was weighed for TSS quantification, the paper and tin containing the retained matter was then ignited at 550 °C for more than 20 minutes and then cooled in a dessicator for at least 30 minutes, and then weighed. The VSS for each sample was calculated as the weight of the dried filter minus the weight of the ignited filter divided by the sample volume as shown on Equation 3.7 below.

$$\text{mg VSS/L} = \frac{((\text{weight of dried filter}) - (\text{weight of ignited filter})) \times 1000}{\text{sample volume in mL}} \quad (3.7)$$

3.6.2 Preparatory Procedures and Analysis of Chemical Oxygen Demand (COD) from Wastewater Samples

The total COD concentration was measured using the reactor digestion method and HACH Medium Range (0 to 1500 mg/L COD) reagent vials. The reactor digestion method, using the HACH Low Range (0 to 150 mg/L COD) reagent vials, was used (DR/400, 594) for soluble COD concentration measurements.

A 2-mL of wastewater sample was added to the HACH COD vial. The vial was capped and inverted several times to mix the reagent and wastewater. The COD vials were then heated to 150° C in a heating block for two hours. At the end of two hours, the heater was turned off and the vials were allowed to cool for 20 minutes. The vials were then inverted several times to mix and the glass surface was cleaned. A blank sample prepared exactly as the sample except with deionized water instead of wastewater. For the 0-150 mg/l COD vials, the HACH Spectrophotometer was set on program 2710 (620-nm wavelength) and zeroed with the blank. For 0-1500 mg/l samples, the HACH Spectrophotometer was set on program 2720 (420-nm wavelength) and zeroed with the blank. The COD of the sample was determined using standard COD concentration versus absorbance curve at the program wavelength.

3.6.3 Preparatory Procedures and Analysis of Ammonia-Nitrogen from Wastewater Samples

Ammonia-nitrogen concentration in the wastewater samples was measured using the Salicylate method (HACH Reagents). Two Am Ver Diluent Reagent Test 'N' Tubes were used in this method. 2.0 mL of sample is added to one vial and 2.0 mL of deionized water to the other vial (blank). One Ammonia Salicylate Reagent Powder Pillow was added to each vial. Next, one Ammonia Cyanurate Reagent Powder Pillow was also added to each vial. Vials were then tightly capped and inverted vigorously to dissolve the powder. After the required 20-minute reaction period the blank and sample were cleaned. The blank was used to zero HACH program 2460. The sample was then placed in the DR/4000 Spectrophotometer to determine the ammonia-nitrogen concentration using a standard ammonia-nitrogen concentration versus absorbance (655 nm wavelength) curve. The estimated LOD for this method is 0.031 mg/L NH_3-N. The range of this test is 0 to 2.500 mg/L NH_3-N.

3.6.4 Preparatory Procedures and Analysis of Nitrite-Nitrogen from Wastewater Samples

Nitrite-nitrogen was measured using the Diazotization Method (HACH Reagents). A HACH sample cell was filled with 10 mL of sample. The contents of one Nitra Ver 3 Nitrate Reagent Powder Pillow is added, the cell capped, and the cell is shaken vigorously to dissolve the pillow contents. A 20-minute reaction time is required before analyzing the sample. A blank cell is prepared using 10 mL of deionized water. HACH program 2610 is selected and zeroed with the blank. The sample is then analyzed with the DR/400 Spectrophotometer to obtain the nitrite-nitrogen concentration using a standard nitrite-nitrogen concentration versus absorbance (490-nm wavelength) curve. The estimated LOD for this method is 0.0008 mg/L NO_2-N. The range for this method is 0 to 0.3000 mg/L NO_2-N.

3.6.5 Preparatory Procedures and Analysis of Nitrate-Nitrogen from Wastewater Samples

Nitrate-nitrogen was measured using the Cadmium Reduction Method (HACH Reagents). A HACH sample cell was filled with 10 mL of sample. The contents of one Nitra Ver 5 Nitrate Reagent Powder Pillow is added, the cell capped, and the cell is shaken vigorously to dissolve the pillow contents. A blank cell is prepared using 10 mL of deionized water. HACH program 2530 is selected and zeroed with the blank. The sample is then placed in the DR/4000 Spectrophotometer to determine the nitrate-nitrogen concentration using a standard nitrate-nitrogen concentration versus absorbance (500-nm wavelength) curve. The estimated LOD for this method is 0.5 mg/L. The range of detection for this method is 0 to 30.0 mg/L.

3.6.6 Oxidation-Reduction Potential (ORP) and pH

Both ORP and pH were measured using the same device. A Beckman 200 series meter and Corning G-P Combo electrode was used. The 220-model meter and electrode were calibrated using VWR standard buffer pH 4.0 and pH 10.0 solutions before each use. The meter provide real time pH and mV measurements. ORP could be measured between ±999 with an accuracy of ±0.2 mV or 0.2T the relative mV reading, whichever is greater. Sample and experimental pH could be measured between 0 and 15.99 pH units with an accuracy of ±0.01 pH units.

The pH of the reactor was also routinely measured using pH paper. Hydrion paper, 6.0 to 8.0 range, was used. The paper color was compared to the color comparator chart. This method provided the pH of the wastewater within 0.2 pH units and was used to verify the pH meter results.

3.6.7 Dissolved Oxygen (DO)

DO levels were measured with a YSI Model 58 Dissolved Oxygen Meter using a YSI Model 5739 probe. Samples were placed in a BOD bottle, a Teflon stirrer bar was added, and the bottle was capped with the probe. A magnetic stirrer continuously mixed the sample throughout the measurement period. The meter scale of 0.01 mg/L was used for DO measurement. The temperature could also be measured between 5 °C and 45 °C.

3.6.8 Alkalinity

The titration method (standard methods section 2320B) was used to determine the total alkalinity of samples. Samples were first warmed to room temperature. A given volume of the sample was then transferred to a 250-mL beaker (usually 100 or 200 mL sample using a volumetric flask). A pre-made solution of 0.02 N H2SO4 from Baker Chemical

Company was used to titrate the sample to the pH endpoint of about 4.3. The pH was monitored with the Beckman 220 meter and Corning electrode. Total alkalinity (ALK) as mg/L of $CaCO_3$ as follows:

$$ALK = \frac{(\text{Acid Added, mL}) * (\text{Normality, meq/L}) * 50000 \text{ mg } CaCO_3 / \text{meq}}{(\text{sample volume, mL})} \quad (3.8)$$

3.6.9 Phosphorus

Phosphorus was measured during anoxic reactor experiments for each WWTP to check for check ortho-phosphorus release indicating the presence of PAOs during field-testing. Orthophosphate concentrations were measured using the Ascorbic Acid Method and HACH Chemical Company Reagents.

A HACH sample cell was filled with 10 mL of sample. The contents of one Phos Vera 3 phosphate Reagent Powder Pillow is added, the cell capped, and the cell was shaken vigorously to dissolve the pillow contents. A two-minute reaction period is allowed. A blank cell is prepared using 10 mL of deionized water. HACH program 3025 is selected and zeroed with the blank. The sample is then placed in the DR/4000 Spectrophotometer and analyzed for PO_4^{3-} using a standard concentration versus absorption curve for ortho-phosphorus (890-nm wavelength). The estimated LOD for this method is 0.045 mg/L PO_4^{3-}. The range of detection for this method is 0 to 2.500 mg/L PO_4^{3-}.

Chapter 4 Results and Discussion

This chapter presents the results of the field experiments conducted, parameters used to evaluate the denitrification kinetics observed, and discuss these results. \The experimental results from the anoxic reactor site tests and the laboratory denitrifying biomass fractionation tests are presented and discussed for each WWTP, and then all the plant results are compared and reviewed in a subsequent section.

4.1 Olympic Terrace Results

4.1.1 Trial Anoxic Reactor Experiments Conducted at Olympus Terrace

The first field experiments were conducted at Olympus Terrace in July 1998. These experiments were considered preliminary and were performed to test the procedures and time required for the site work. During the testing the site reactor was operated at different HRTs for data collection on specific nitrate removal rates as a function of reactor RBCOD and TSBCOD concentrations. Each anoxic zone test HRT condition used was defined as a test run. Initially, four different HRTs were to be evaluated without acetate addition and four HRTs were to be evaluated with acetate addition resulting in a total of eight runs during each set of field site tests. Due to time limitations the experimental runs were reduced to six runs, three with and three without acetate addition. The preliminary tests also showed that the size of the nitrogen gas-tank (20 CF of N_2 gas) used initially was too small and thus could not provide nitrogen sparging for the entire duration of the field experiments. A larger 60 CF N_2 gas source was used and proved adequate for future site tests. Acetic acid was initially used for acetate addition, but the wastewater at did not provide an adequate buffering capacity and the pH dropped from about 7.0 to less than 5.0 with the addition 30 mg/L of acetic acid. To avoid on-site pH manipulation as a result of acetic acid addition, sodium acetate was used in future tests.

During the preliminary laboratory endogenous nitrate utilization tests conducted in July, a 30-40 minute lag time before denitrification was observed. This may have been related to the cold temperature storage of the mixed liquor or indicated that the biomass required an acclimation period to the anoxic conditions before beginning endogenous nitrate respiration. The experimental protocol was modified for future site tests to include adding 10 mg/L of nitrate to the RAS container to ensure that the RAS entering the reactor was acclimated to the anoxic conditions.

4.1.2 SDNR's in Anoxic Reactor Experiments at Olympus Terrace

The first set of reliable experimental testing was conducted at the Olympus Terrace WWTP on August 5, 1998. The site test anoxic reactor described in Chapter 3 was setup and operated. The raw data from the six experimental runs conducted are in Appendix 1. The reactor temperatures for these tests ranged from 18.5 to 20.5 °C. ORP measurements started at –10 mV at reactor start up and ranged from –30.9 mV to –47.8 mV at steady state. More negative ORP values were generally associated with the shorter HRTs and higher ORP values (less negative) were associated with longer HRTs. The HRT was controlled by reactor flow rates from the wastewater and RAS feed sources. Runs without acetate addition had more negative ORP readings (approximately –10 mV difference), but the wastewater used during the acetate addition test runs also had a much lower pH than the runs without acetate addition (7.2 with acetate runs and 7.8 without acetate addition runs). Reactor samples were analyzed on August 5-6, 1998.

On September 15, 1998 a second set of experiments were conducted at the Olympus Terrace WWTP. The objective of these additional experiments were to obtain denitrification kinetic information at a longer anoxic reactor HRT and lower reactor soluble COD concentrations. The anoxic site reactor temperatures were slightly lower during these experiments, ranging from 17 to 19.3 °C. Site reactor ORP measurements ranged from –46.1 mV to –61.1 mV and were more negative at the longer HRTs.

The raw data for all experiments conducted at Olympus Terrace are located at Appendix 1. A summary of important test site influent conditions and reactor conditions at steady state for each experimental run is shown in Tables 4.1 and 4.2, respectively.

Table 4.1 Olympus Terrace Site Test Reactor HRT and Influent Substrate Concentrations

Run #	HRT (min)	TBSCOD* (mg/L)	RBCOD* (mg/L)	NO_3-N (mg/L)	Acetate Addition
1	25.0			15.7	N
2	19.0			16.1	N
3	13.4			14.8	N
4	29.6			14.1	Y
5	20.5			14.8	Y
6	14.4			16.8	Y
7	38.8	60.6	44.8	23.1	N
8	14.4	42.5	30.5	22.9	N
9	37.8	200	153.8	24.7	Y
10	15.1	181	146	25.1	Y

*Influent CODs for runs 1 to 6 were not reliable due to sample storage issues and are thus not reported.

Table 4.2 Olympus Terrace Site Test Reactor Conditions and Reactor Substrate Concentrations

Run #	MLVSS (mg/L)	Temperature (°C)	pH	TBSCOD (mg/L)	RBCOD (mg/L)	NO_3-N (mg/L)
1	1489	18.5	7.47	28.2	3.2	12.6
2	1476	18.8	7.54	40.2	6.7	14.5
3	1145	19.3	7.52	49.2	10.6	13.9
4	1421	20.0	7.33	59.2	15.7	9.2
5	1414	20.5	7.22	73.2	37.4	11.2
6	1210	20.1	7.18	78.2	41.1	14.5
7	1550	17.0	7.76	12.9	14.4	18.0
8	1827	17.9	7.56	22.2	23.1	20.6
9	1985	18.3	7.84	153.6	127.2	16.4
10	1983	19.3	7.75	158.8	141.0	21.3

The soluble CODs shown are biodegradable soluble CODs since the plant effluent flocculated SCOD, which represents the non-biodegradable SCOD, was subtracted from all sample flocculated soluble CODs and soluble CODs, respectively. The RBCOD and TBSCOD was determined using Equation 4.1 and 4.2.

$$RBCOD = FCOD - Effluent\ FCOD \qquad (4.1)$$

$$TBSCOD = SCOD - Effluent\ SCOD \qquad (4.2)$$

The first kinetic parameter of interest was the specific denitrification rate (SDNR) as this has been used before for anoxic reactor designs and there is literature available for comparison. The SDNRs are shown as a function of the test HRT and then as function of the reactor SCOD concentrations. The observed specific denitrification rates (SDNRs) for each test run were determined from the steady state test results using Equation 4.2 below:

$$SDNR_{OBS}\ (g/g\text{-}day) = \frac{(nitrate_{IN} - nitrate_{OUT} - mg/L) * (1440 - min/day)}{(reactor\ VSS\ concentration - mg/L) * (HRT - min)} \qquad (4.3)$$

The observed SDNR versus HRT for all site reactor test runs conducted at Olympus Terrace is shown in Figure 4.1.

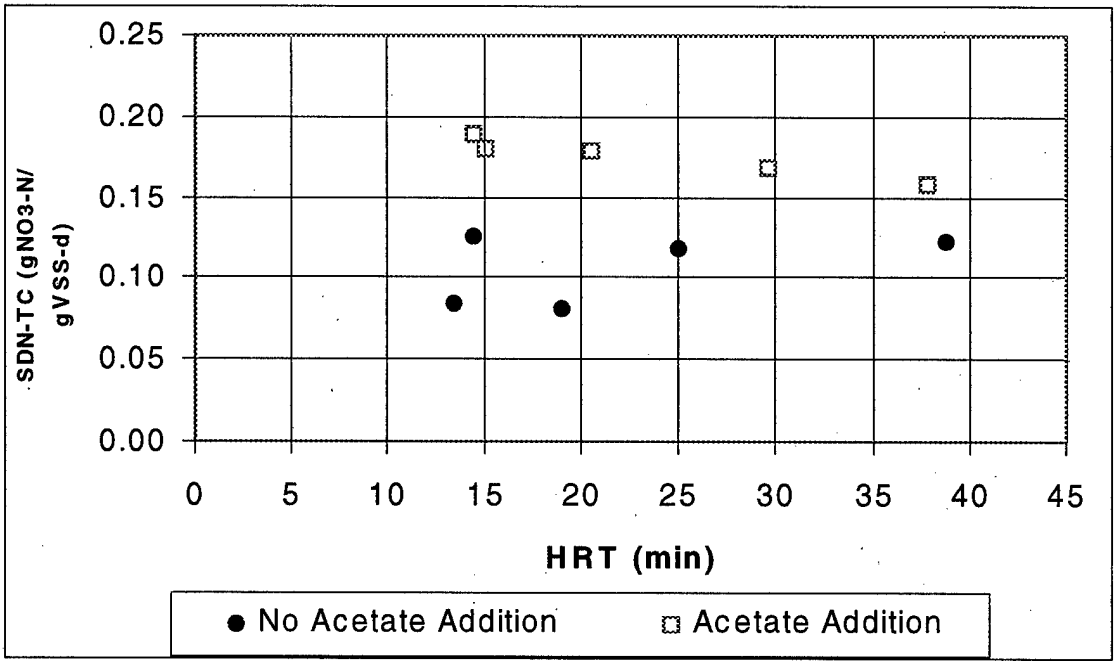

Figure 4.1 Observed SDNR (g NO_3-N/g VSS-d) versus HRT(min) for Olympus Terrace Test Runs

The SDNR varied over a wide range at the lower HRTs and no clear correlation between SDNR and HRT is seen. The SDNR values of 0.08 to 0.19 mg NO_3-N/mg VSS-d are within the range of reported literature values.

Based on the fundamental kinetics the SDNRs should be related to the reactor soluble degradable COD concentration with higher rates (higher SDNRs) with higher soluble COD concentrations. Figures 4.2 and 4.3 compare the SDNRs with the reactor TBSCOD and RBCOD concentrations.

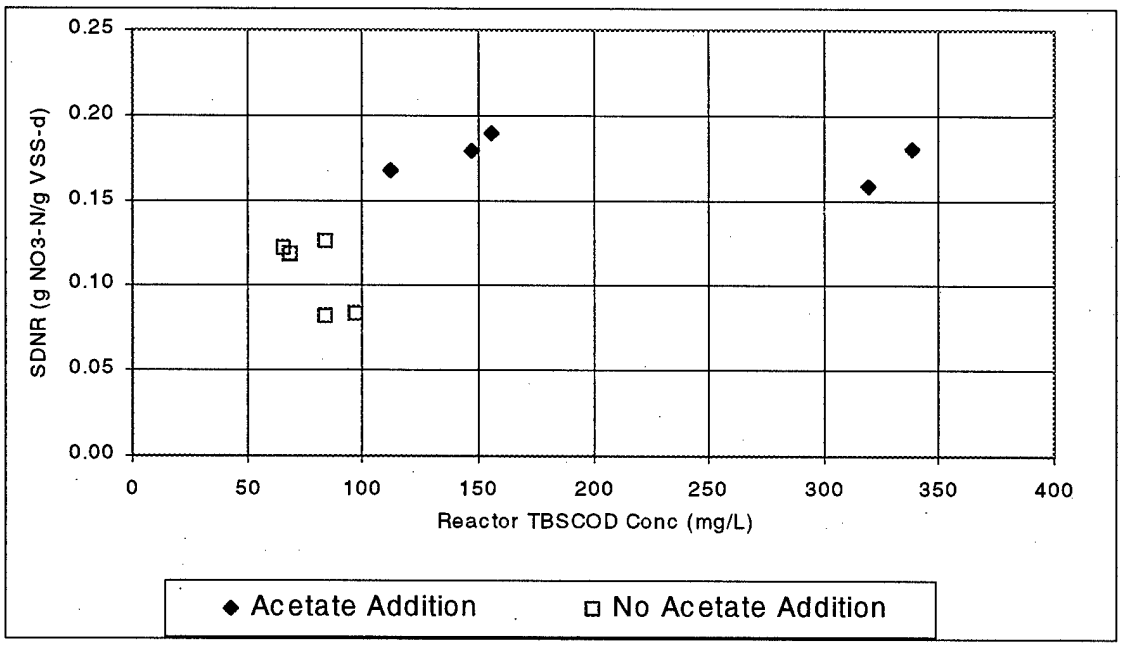

Figure 4.2 Observed SDNR (g NO3-N/gVSS-d) versus Reactor TBSCOD Concentration (mg/L) for all Olympus Terrace Test Runs

Figure 4.2 shows that the SDNR generally increases as a function of reactor TBSCOD concentration. The higher values were associated with higher reactor TBSCOD concentration due to acetate addition. Between reactor TBSCOD concentrations of 60 to 100 mg/L there was no clear trend between SDNR and TBSCOD.

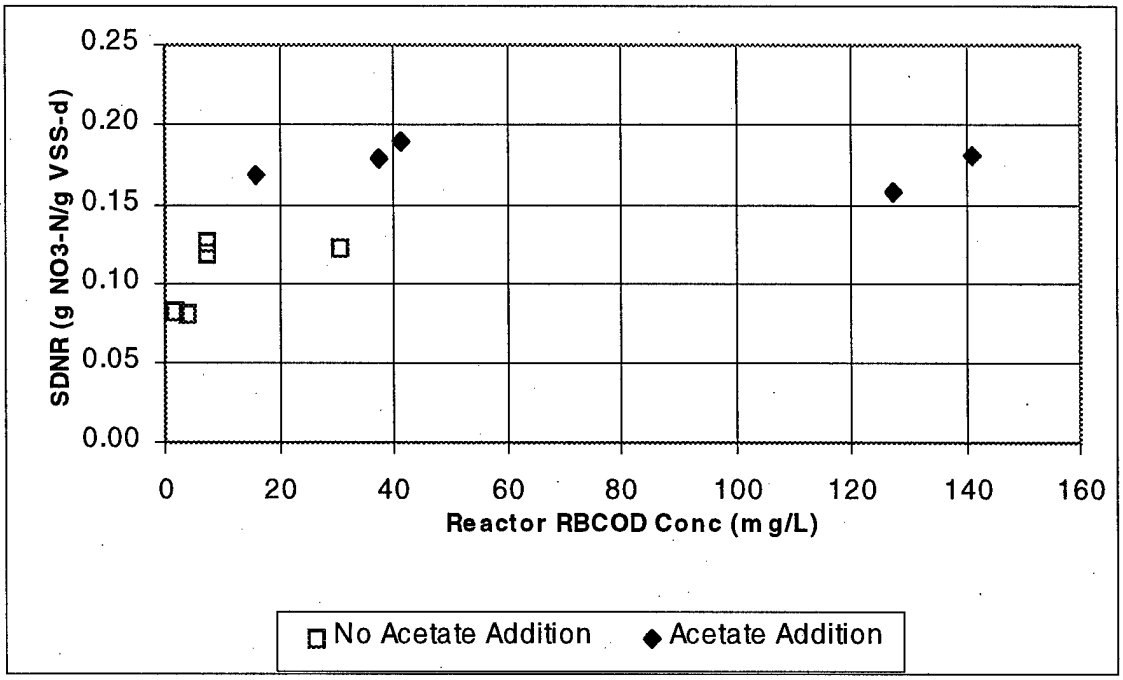

Figure 4.3 Observed SDNR (g NO3-N/gVSS-d) versus Reactor RBCOD Concentration (mg/L) for all Olympus Terrace Site Tests

The SDNRs followed a similar pattern with reactor RBCOD concentration as observed with the TBSCOD concentration. A more general trend of increasing SDNR with increasing reactor RBCOD concentration is seen at the lower RBCOD concentrations with the exception of the one data point at the 30 mg/L reactor RBCOD concentration.

To compare SDNRs obtained at different site reactor temperatures, the SDNRs were corrected to 20 °C using Equation 2.3 with the BioWin model temperature value (θ = 1.029). The temperature corrected SDNR represents a small change from the uncorrected values since the test reactor temperature ranged from 17 to 21 °C. The corrected SDNRs (SDNR-TC) are shown in Figures 4.4 and 4.5 as a function of reactor TBSCOD and RBCOD concentration, respectively.

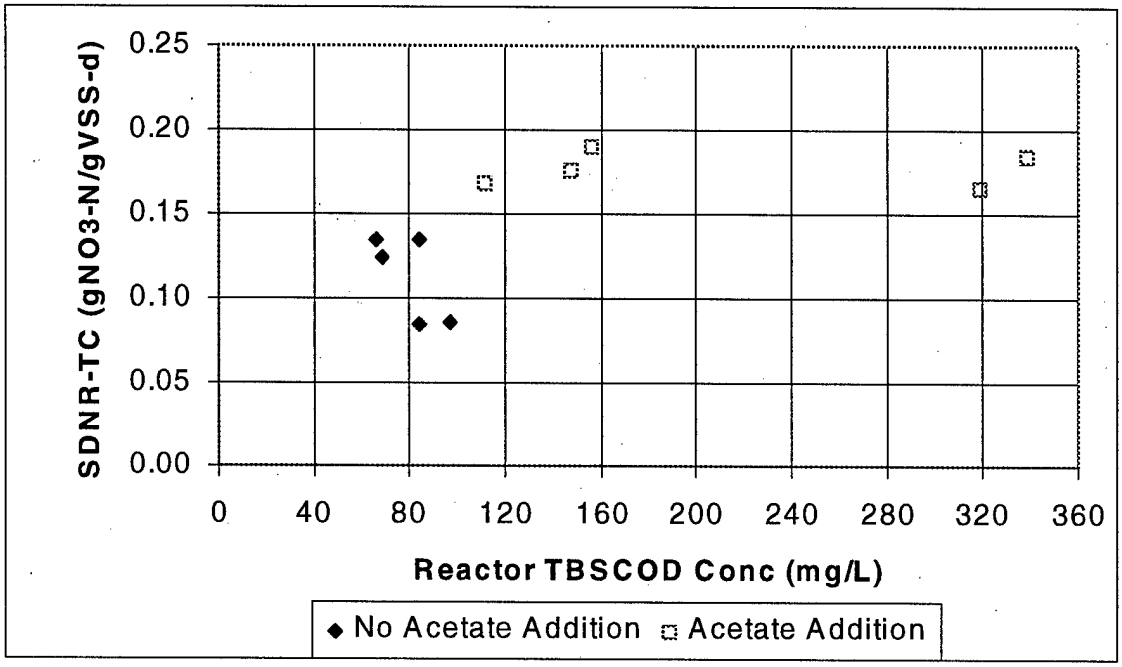

Figure 4.4 $SDNR_{TC}$ (g NO3-N/gVSS-d) versus Reactor TBSCOD Concentration (mg/L) for all Olympus Terrace Test Runs

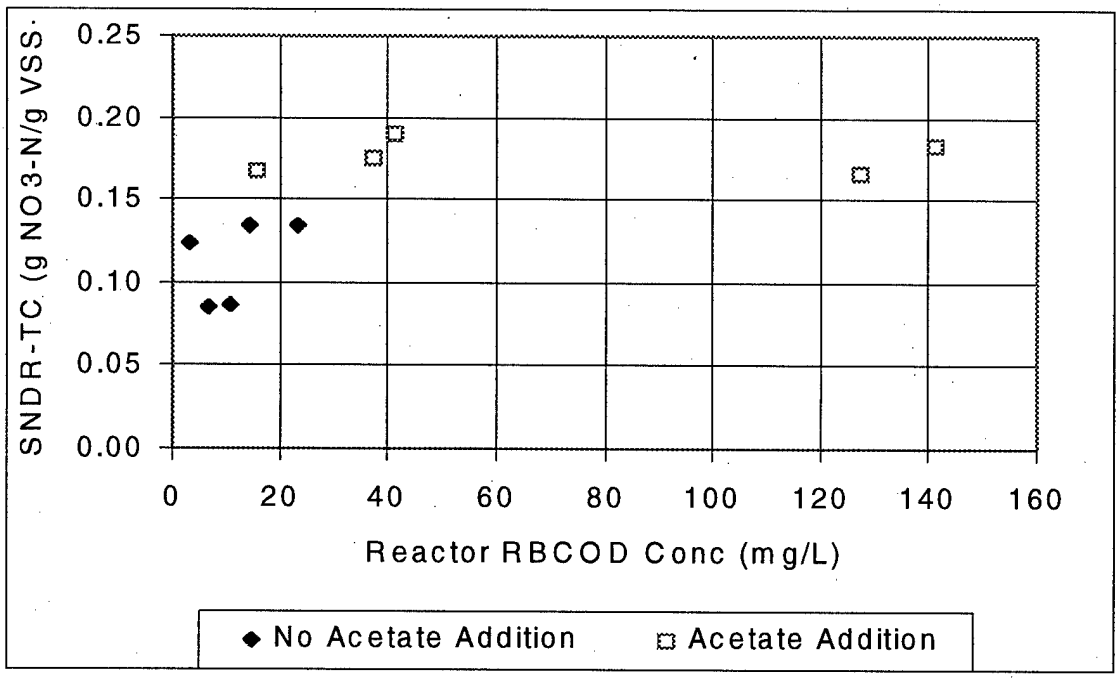

Figure 4.5 $SDNR_{TC}$ (g NO3-N/gVSS-d) versus Reactor RBCOD Concentration (mg/L) for all Olympus Terrace Test Runs

Observed SDNR rates ranged from 0.085 to 0.202 g NO_3/g VSS-d after correcting the SDNR to an equivalent rate expected at 20 °C. The temperature correction resulted in slightly higher rates for all but the three runs with temperatures greater than 20 °C. The temperature corrected SDNRs show a stronger trend with the reactor RBCOD concentrations than for the SDNR versus reactor RBCOD relationship without the temperature correction. The SDNR trend with RBCOD appears to be more obvious then with SDNR versus TBSCOD concentrations.

The data was examined to see how well the observed SDNRs fit the SDNR versus F/M ratio relationship by Burdick et al. (1982). The F/M ratio in the Burdick et al. (1982) relationship is based on total influent BOD_5 and a MLSS concentration. For Figure 4.5 this had to be converted to an F/M ratio on the basis of a biodegradable soluble COD and MLVSS concentration. This was done by assuming an MLVSS/MLSS ratio of 0.85, and total COD /total BOD_5 ratio of 1.5, and that 40% of the total COD was soluble biodegradable COD. Thus the SNDR versus F/M equation value used in Figure 4.5 is given by Equation 4.4 where F/M is in g TBSCOD/g MLVSS-d.

$$SDNR = (0.0588 \times F/M) + 0.029 \qquad (4.4)$$

The observed SDNR-TC versus F/M ratios for experimental runs 7 to 10 are shown in Figure 4.6 (runs 1-6 were not included due to lack of reliable influent COD values).

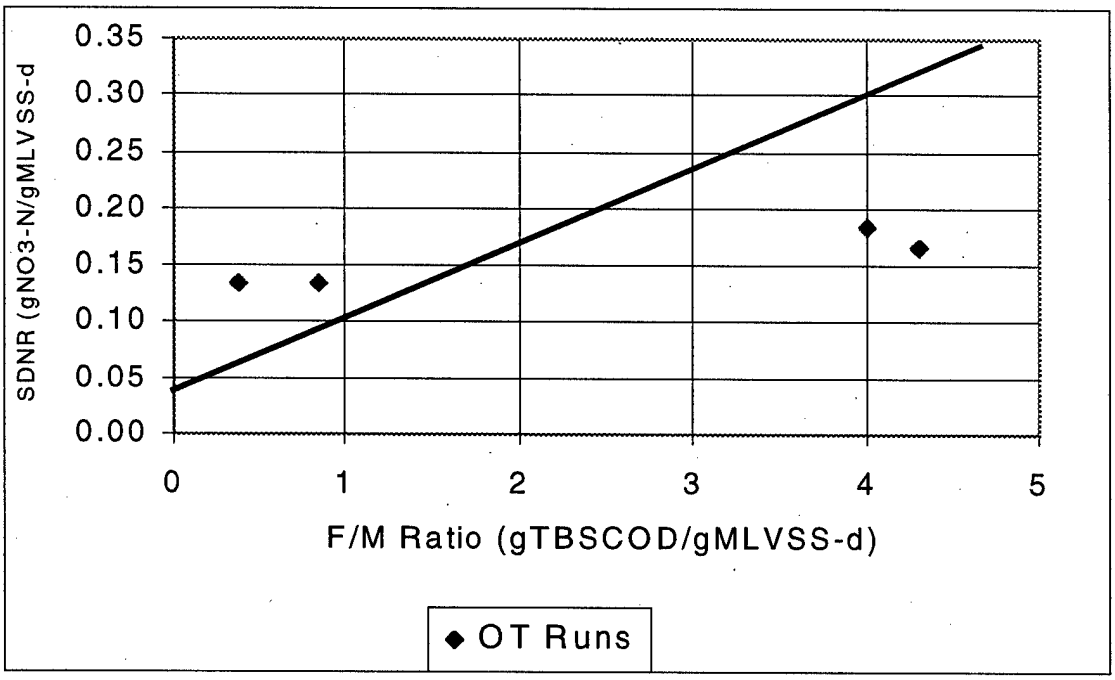

Figure 4.6 Observed SDNR (g NO3-N/gVSS-d) versus F/M Ratio (gTBSCOD/gVSS-d) for Site Reactor Test Runs 7-10 Conducted at Olympus Terrace

The F/M equation is represented by the solid line in Figure 4.6. Observed SDNRs increased slightly with F/M Ratio for runs 7-10, but the trend does not match that predicted by the F/M equation.

4.1.3 Adjustment of SDNR for Estimated Active Biomass

The observed SDNRs shown in section 4.1.2 are based on the measured MLVSS concentrations, which includes volatile solids other than active biomass. These can be inactive biomass resulting from cell decay, non-biodegradable influent VSS, and biodegradable VSS remaining. The fraction of active biomass may vary from plant to plant depending on the amount of non-degradable influent VSS entering the plant (plants without primary treatment will have a higher influent concentration of nondegradable VSS) and the activated sludge operating SRT. To compare the SDNRs from these tests to the results of tests conducted at other plants, the SDNR values were normalized to an

active biomass concentration estimated from the measured test MLVSS concentration and plant data that related to solids production and BOD removed. The active biomass fraction (AVSS) is given by Equation 4.5.

$$AVSS = X_a / X_T \qquad (4.5)$$

Where: AVSS = Estimated fraction of active biomass in MLVSS

X_a = Estimated active biomass concentration, mg/L

X_T = measured plant MLSS concentration, mg/L

The aeration tank active biomass concentration was estimated by the following equation from Metcalf & Eddy (1991) using a value of 0.6 g VSS/g BOD_5 removed for Y and 0.08 g/g-d for b.

$$X_a = \left(\frac{Y_{bio}}{1+b*SRT}\right) * \left(\frac{Q \times \Delta BOD \times SRT}{V}\right) \qquad (4.6)$$

where: Q = Average influent plant flow, MGD

SRT = Plants aerobic mean cell residence time, days

V = Aerobic zone volume

ΔBOD = BOD removed, plant effluent BOD minus the plant influent BOD (or primary effluent BOD if plant has primaries), mg/L

From this the amount of active biomass wasted per day from the plant could be estimated based on the average SRT of the plant.

$$Px_{bio} = (X_a * V) / SRT \qquad (4.7)$$

Where: Px_{bio} = Active biomass wasted per day, lb/day

The average daily solids wasting rate was provided for each plant and is defined as follows:

$$Px_T = \frac{(X_T)*(V)}{SRT} \qquad (4.8)$$

The active biomass wasted per day is also proportional to the mixed liquor active biomass concentration and SRT:

$$Px_{BIO} = \frac{(X_a) * (V)}{SRT} \qquad (4.9)$$

Using the plant data for Px_T and the Px estimated , Px_{BIO} values calculated based on the plant average BOD removal and SRT, the active biomass fraction can be estimated.

$$AVSS = \frac{Xa}{X_T} = \frac{Px_{BIO}}{Px_T} \qquad (4.10)$$

Where Px_T = Average mass of solids wasted per day, lb/day

The following plant operating data was provided from the Olympus Terrace WWTP for the month of August and was used to calculate the estimated active biomass fraction of the mixed liquor.

Table 4.3 Olympus Terrace WWTP Operating Data for August 1998

Parameter	*Value*
SRT, days	13
Average Flow, MGD	1.35
Influent BOD, mg/L	192
Effluent BOD, mg/L	5.1
Solids Wasted per Day, lb/day	1547.8

A comprehensive listing of Olympus Terrace's general operating information is available at Appendix 1.

The net active biomass yield for Olympus Terrace is determined using Equation 4.11 as:

$$Y_{bio} = Y / (1+b*SRT) \qquad (4.11)$$

$$Y_{bio} = (0.6)/(1+(0.08)*(13)) = 0.29 \text{ g TSS/g BOD}$$

The pounds of biomass produced per day is estimated using equation 4.12.

$$Px_{Bio} = Y_{bio} * (\Delta BOD)*Q \qquad (4.12)$$

$$P_{XBio} = (0.29\ g/g)*(192 - 5.1\ mg/L)*(1.35\ MGD)*8.34 = 610.3\ lb/d$$

Since the reported plant average sludge wasting rate was 1547.8 lb/d for August, the active biomass fraction is estimated using Equation 4.10 as:

$$AVSS = (610.3)/(1547.8) = 0.39$$

Thus, the estimated active biomass is 39% of the plant's MLVSS concentration.

As a check on the reasonableness of this estimation, the theoretical active biomass was determined using general relationships as outlined by Randall et al. (1992).

$$AVSS_{theo} = Y_{bio} / (Y_{bio}+Y_{inerts}) = \frac{\left[\frac{Y}{(1+b*SRT)}\right]}{\left[\frac{Y}{(1+b*SRT)} + Y_i\right]} \qquad (4.13)$$

Where Y_{inerts} = 0.4 - 0.5 g TSS/g BOD for WWTP without primary treatment

Y_{inerts} = 0.2 – 0.3 g TSS/g BOD for WWTP with primary treatment

Inert yields for WWTPs without primaries, like Olympus Terrace, were reported to vary from 0.4 to 0.5 grams of inerts per gram of BOD removed. Using an inert yield of 0.5 the active biomass fraction is estimated:

$$AVSS = 0.29\ g\ VSS/g\ BOD / (0.29\ g\ VSS/g\ BOD + 0.5\ g\ TSS/g\ BOD) = 0.36$$

Since the estimated AVSS of 0.39 based on the plant data is close to the general theoretical AVSS of 0.36, the estimated AVSS based on the plant data is considered to be a reasonable estimate. This comparison provides a check on the reasonableness of the plant data.

The SDNR based on active biomass was determined by dividing the temperature corrected SDNR by the estimated active mass fraction of the MLVSS. These AVSS corrected SDNRs ($SDNR_{AVSS}$) varied from 0.21 to 0.508 g NO_3-N/g AVSS-d and are shown in

Figures 4.7 & 4.8 as a function of the reactor TBSCOD and RBCOD concentrations, respectively.

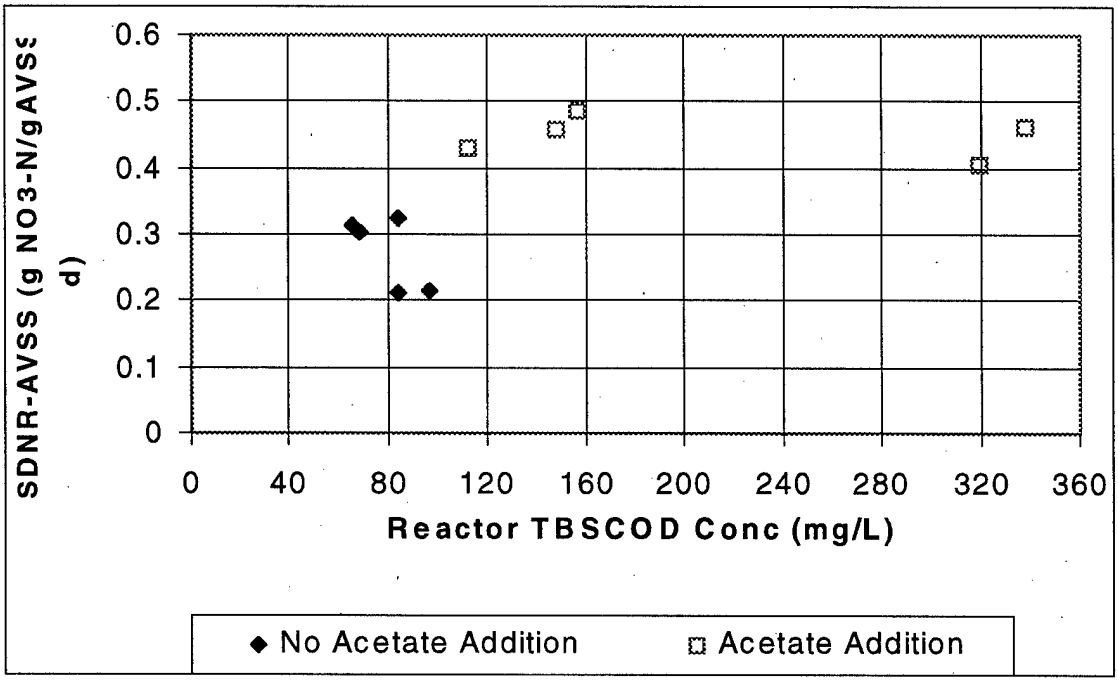

Figure 4.7 $SDNR_{AVSS}$ (g NO3-N/gAVSS-d) versus Reactor TBSCOD Concentration (mg/L) for all Olympus Terrace Test Runs

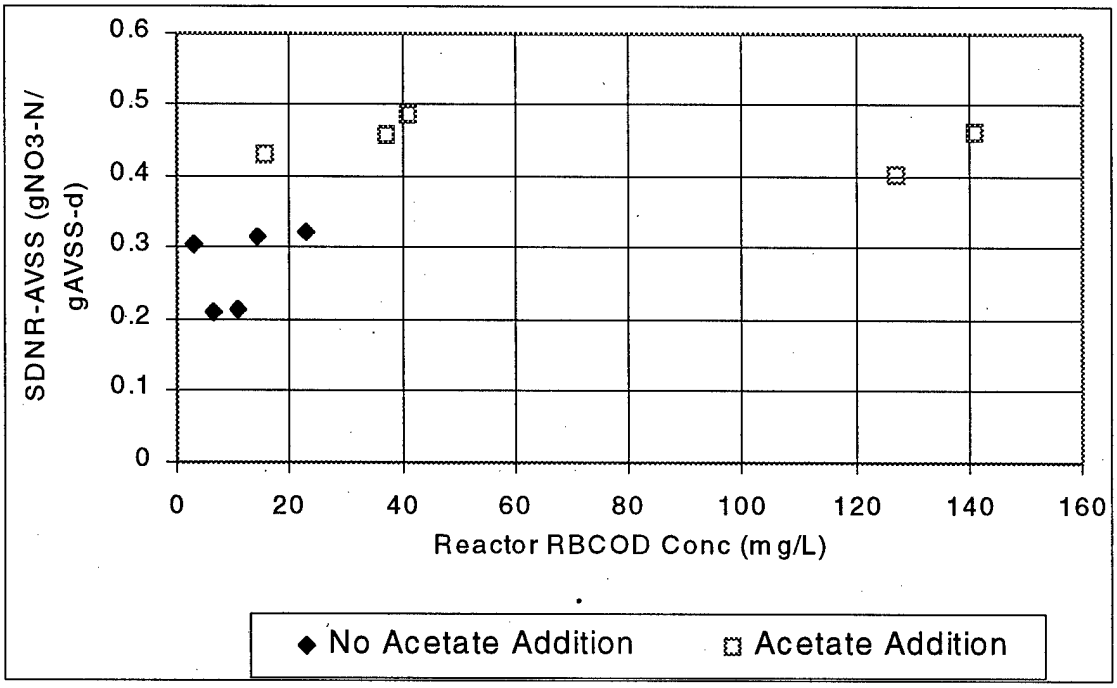

Figure 4.8 $SDNR_{AVSS}$ (g NO3-N/gAVSS-d) versus Reactor RBCOD Concentration (mg/L) for all Olympus Terrace Test Runs

The active mass fraction correction resulted in a 138% increase in SDNR values. After the active biomass fraction correction, the SNDRs show a general increase with increasing RBCOD and TBSCOD reactor concentrations.

4.1.4 Endogenous Respiration Batch Test Results

Not all of the heterotrophs are expected to be facultative and thus consume both oxygen and nitrate-nitrogen as electron acceptors. A comparison of equivalent oxygen consumption rates under endogenous conditions with the same biomass under both aerobic and anoxic conditions was used to indicate the fraction of the active biomass capable of using nitrate-nitrogen as an electron acceptor.

The mixed liquor endogenous uptake tests were conducted on August 6, 1998. After four hours of aeration 15 mg/L of nitrate was added to the mixed liquor and the Endogenous

Oxygen Utilization Rate (OUR_{endog}) test was conducted as discussed in Chapter 2. The results of the OUR_{endog} test are shown in Figure 4.9.

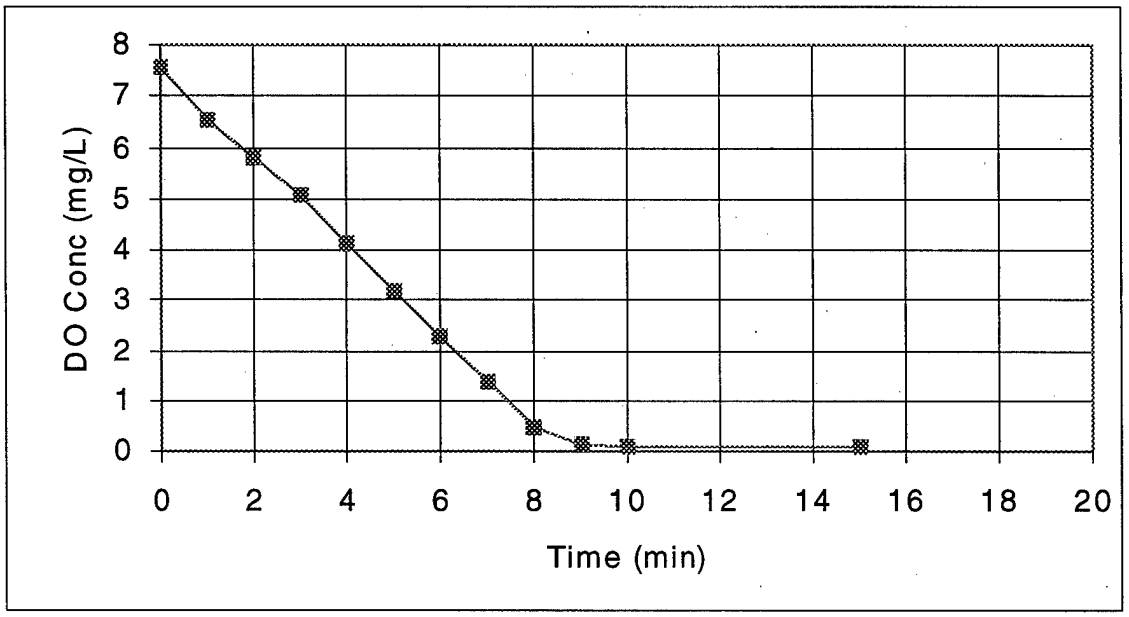

Figure 4.9 Olympus Terrace OUR_{endog} Test Results
(MLVSS = 4100 mg/L, Temp = 16.6 °C)

Using the Microsoft Excel SLOPE function, the slope of the data over the linear oxygen utilization range was determined as -1.0 mg O_2/L-min with an endogenous OUR of 60 mg O_2/L-hr. The Specific Endogenous Oxygen Utilization Rate (SEOUR) was determined by dividing the OUR by the MLVSS concentration of 4100 mg/L. The MLVSS concentration was based on the analysis of two mixed liquor samples taken at the end of the OUR test. The resulting SEOUR for the test was 0.29 g O_2/gVSS-d. The temperature during the test was 16.6 °C. The SEOUR rate was corrected to 20 °C using Equation 2.3 to give an $SOUR_{20}$ of 0.32-g O_2/gVSS-d. Using the 1.42 g O_2/g MLVSS equivalency ratio, the MLVSS decay coefficient was estimated as 0.23 g/g-d. This decay coefficient is much larger than the value reported in the literature and 0.08 value used in BioWin. The

sample storage may have affected the results or aeration time period may not have been long enough to consume all of the available substrate within the mixed liquor sample.

Next, the endogenous nitrate utilization rate (NUR_{endog}) was determined for the same mixed liquor as outlined in Chapter 2. The literature review (Payne and Riley (1969)) indicated that after long periods of aeration the nitrate reductase enzyme is no longer produced. A lag time of 40 minutes occurred (Figure 4.9) before nitrate reduction proceeded at a fairly linear rate. The 24 hours of storage, followed by the four hours of aeration to before the NUR test was conducted may have been long enough to produce a nitrate reductase enzyme deficiency. The results of the NUR_{endog} test are shown in Figure 4.10 below:

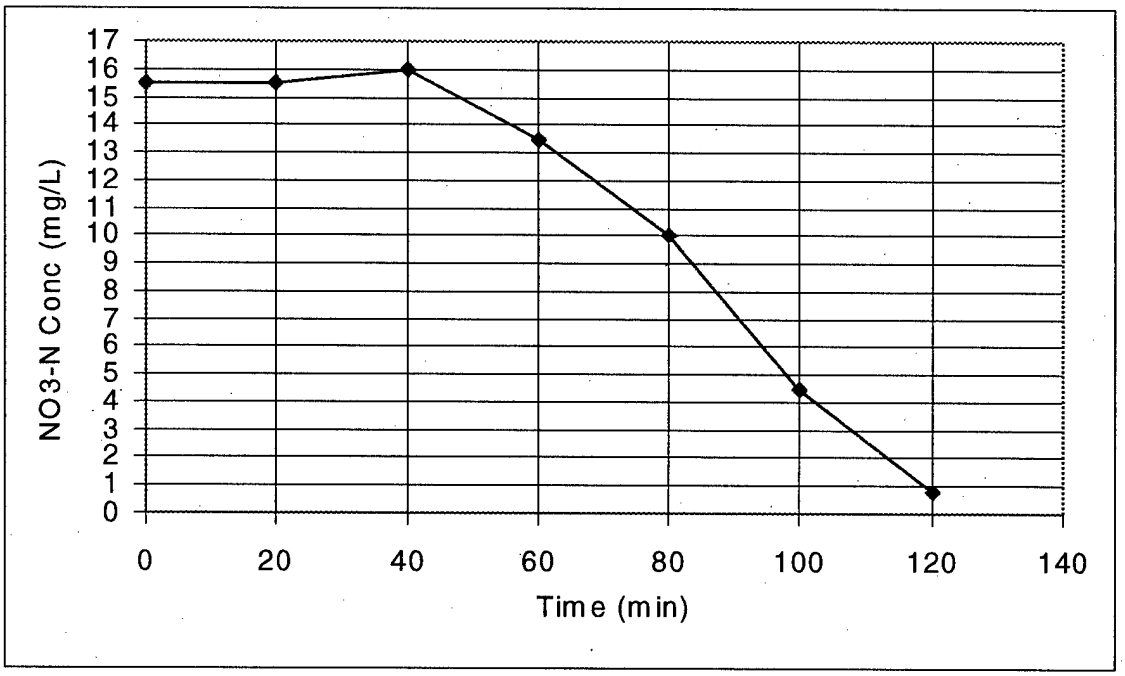

Figure 4.10 Olympus Terrace NUR_{endog} Test Results
(MLVSS = 4100 mg/L, Temp = 23.9 °C)

Using the Microsoft Excel SLOPE function, the slope of the data over the linear nitrate utilization range was -0.16 mg NO_3-N/L-min. The $SDNR_{endog}$ was then calculated to be 0.07 g NO_3-N/gVSS-d. The temperature during the test was 23.9 °C. The $SDNR_{endog}$ was

corrected to 20 °C using Equation 2.3, to give an $SDNR_{endog20}$ of 0.0584 g NO_3-N/gVSS-d. This results in an equivalent specific oxygen uptake rate of (0.0584 g NO_3-N/gVSS-d)*(2.86 g O_2/g NO3-N) = 0.167 g O_2/gVSS-d. The fraction of active biomass capable of nitrate reduction is thus estimated to be (0.167 g O_2/gVSS-d)/(0.32 g O_2/gVSS-d) or 0.52.

The SDNR site test results was further modified to express the SDNR relative to the biomass that could use nitrate by dividing the $SDNR_{AVSS}$ by the fraction of denitrifying biomass. The resulting SNDR represents nitrate removal by the estimated active biomass capable of denitrification. The active denitrifying biomass SDNR ($SDNR_{ADVSS}$)versus reactor TBSCOD and RBCOD concentrations are shown in Figures 4.11 & 4.12, respectively.

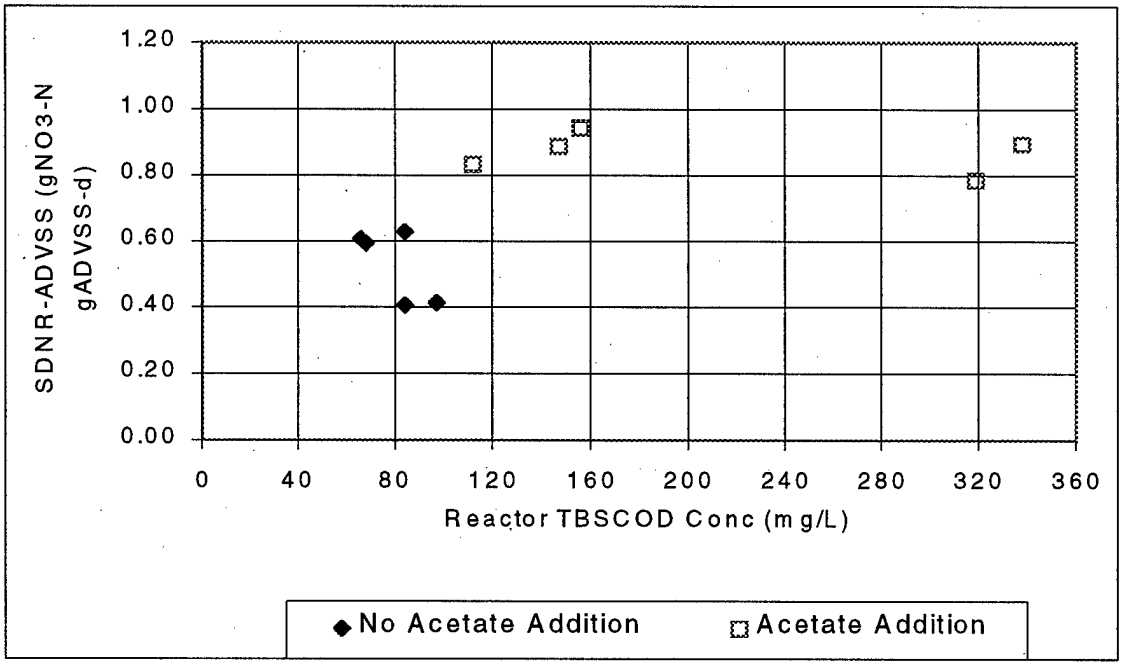

Figure 4.11 $SDNR_{ADVSS}$ (g NO3-N/gADVSS-d) versus Reactor TBSCOD Concentration (mg/L) for all Olympus Terrace Test Runs

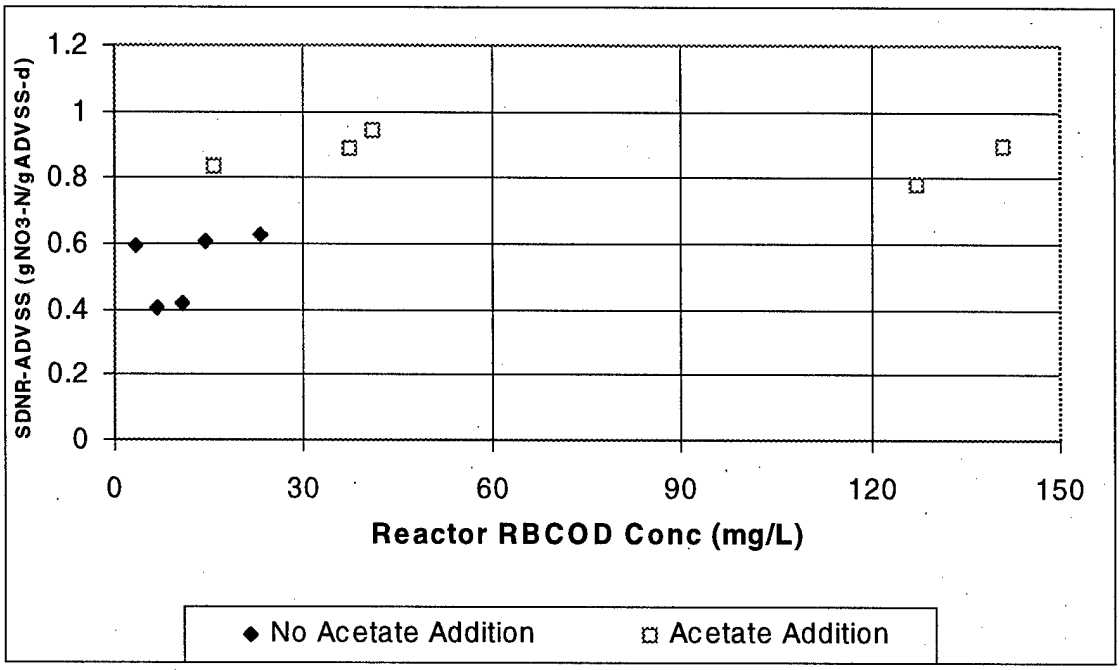

Figure 4.12 $SDNR_{ADVSS}$ (g NO3-N/gADVSS-d) versus Reactor RBCOD Concentration (mg/L) for all Olympus Terrace Test Runs

After the active denitrifying biomass fraction correction, the SNDRs continued to generally increase with both RBCOD and TBSCOD reactor concentrations. The $SDNR_{ADVSS}$ varied from 0.406 to 0.983 g NO_3/g ADVSS-d. A maximum SDNR of 0.983 was apparent with the reactor RBCOD concentration. A maximum SDNR was not indicated with the reactor TBSCOD concentration. These specific denitrification rates represent denitrification kinetics for the actual nitrate utilizing bacteria and thus are expected to be comparable to other WWTP denitrification kinetics.

4.1.5 COD and Nitrate Utilization

The IAWQ model predicts anoxic zone nitrate utilization rates in proportion to COD degradation rates that are related to the active denitrifying biomass and degradable COD concentration. This section focuses on the relationship between COD utilization rates and the observed denitrification rates. As discussed in Chapter 2, Ekama et al (1986) found in

the anoxic batch test experiments that the SDNR is dependent on the COD composition. They found that the RBCOD was utilized more rapidly at first and was followed by utilization of the SBCOD. This sequence of utilization generated two sequential denitrification utilization rates. Both SBCOD and RBCOD utilization are expected to occur concurrently in the reactor experiments and according to the IAWQ model (Barker and Dold 1997a,b) the SBCOD would be hydrolyzed to RBCOD, while the bacteria oxidize and consume the RBCOD.

The substrate utilization rates of biomass during anoxic site tests conducted on September 15, 1998 are reported in Table 4.4.

Table 4.4 Olympus Terrace Specific COD Utilization Rates for Test Runs 7-10 (Based on Active Biomass)

Run	HRT	Reactor Soluble COD Concentrations (mg/L)			Specific COD Utilization Rates (gCOD/gXa-d)			Acetate
	(min)	TBSCOD	BSCOD	RBCOD	R_{TBSCOD}	R_{BSCOD}	R_{RBCOD}	Addition
7	38.8	65.7	51.3	14.4	9.27	5.66	3.61	N
8	14.4	83.7	60.6	23.1	7.52	5.51	2.01	N
9	37.8	319.2	192	127.2	6.95	4.42	2.53	Y
10	15.1	338.2	197.2	141	6.48	5.29	1.19	Y

Unreliable quantification of the influent COD fraction concentrations during runs 1 to 6 at Olympus Terrace resulted in their exclusion from the COD utilization analysis.

This table shows that both RBCOD and TBSCOD are consumed simultaneously in the reactor. The TBSCOD utilization rate is actually the total rate of RBCOD utilization assuming that the BSCOD is converted to RBCOD prior to utilization. The TBSCOD specific utilization rate (SUR) is the sum of the BSCOD and RBCOD specific removal rates. The observed COD utilization rates do not correlate with the reactor soluble COD concentrations. The lower specific COD utilization rates were observed at the higher soluble COD concentrations that resulted from acetate addition. The SDNR results in Figure 4.12 showed that higher SDNRs occurred with higher reactor RBCOD concentrations and that acetate addition increased the SDNR. Since the nitrate consumed should be proportional to the COD removal rate the lack of a correlation between the

specific COD removal rate and reactor soluble COD concentration was not expected. The lack of agreement between the specific soluble COD utilization rate and the soluble COD concentrations suggest that the ratio of COD used to nitrate used varied in the experiments. The nitrate used in the reactor was for endogenous respiration and as an electron acceptor for substrate oxidation. An endogenous respiration rate coefficient was assumed since the endogenous respiration lab tests may not have represented the rates during the field testing. An endogenous decay rate of 0.08 g/g-d based on active mass was used. Table 4.5 shows the nitrate used for endogenous respiration and for RBCOD removal.

Table 4.5 Calculated COD/Nitrate Removal Consumption Ratios and Applied F/M Ratios based on Reactor TBSCOD Concentration during Olympus Terrace Test Runs

RUN	HRT	MLSS	$SDNR_{Endog}$ Usage	Observed SNDR	SDNR for	Spec TBSCOD	COD/ NO3	F/M Ratio
				ADVSS	COD rem	Util		
	(min)	(mg/L)	(g/g-d)	(mg/L)	(g/g-d)	(g/g-d)	(g/g)	(g/g-d)
7	38.8	1550	0.0165	0.609	0.593	9.27	15.6	1.45
8	14.4	1827	0.0165	0.625	0.609	7.52	12.3	2.33
9	37.8	1985	0.0165	0.783	0.767	6.95	9.1	3.84
10	15.1	1984	0.0165	0.894	0.878	6.48	7.4	8.7

An estimated COD/NO3-N ratio can be calculated from the estimated active biomass yield of 0.29 g VSS/g COD. The yield as biomass COD is 0.29 (1.42) or 0.41 gram biomass as COD per gram of COD removed. This means that 0.59 g O_2 is used per gram of COD removed. Dividing by the oxygen equivalent of 2.86 g O_2/g NO_3-N yields 0.206 g NO_3-N used, or 4.9 g COD/g NO_3-N.

The COD/NO3-N consumption ratios for runs 7-10 are all much higher than this theoretical value. One explanation for the difference is that a significant portion of the COD removed in the anoxic reaction was not oxidized but went into cellular storage. Substrate uptake and storage has been an accepted removal mechanism for contact-stabilization bioreactors or high F/M loaded activated sludge contact zones. The F/M ratios shown indicate that the anoxic zones experienced a high substrate loading condition.

Specific COD utilization rates also appear to be inversely related to the F/M Ratio. At low F/M ratios the COD utilization rate can be increased by increasing the F/M ratio until the maximum utilization rate is achieved. From Figure 4.11, SDNR values increase over during runs 7-10, indicating increasing biomass nitrate utilization and thus increased COD use as long biomass yield remains the same. This would account for higher TBSCOD utilization at the higher F/M ratio.

The IAWQ model does not account for soluble COD uptake and storage. It determines a nitrate reduction rate as a function of RBCOD oxidation rates that are governed by the reactor RBCOD concentration and Monod kinetics. The model could conceivably predict a higher substrate oxidation rate than what would occur if substrate storage is a significant removal mechanism for soluble COD in short detention time anoxic zones.

4.2 Snoqualmie Falls Results

4.2.1 Results of Anoxic Reactor Experiments Conducted at Snoqualmie Falls

Anoxic reactor experiments were conducted Snoqualmie Falls on August 17, 1998. Earlier tests showed that the RAS was relatively thin so the RAS was concentrated first by settling before it was added to the RAS reservoir for the tests. The raw data from the six experimental runs conducted are located in Appendix 2. The reactor temperatures for these tests ranged from 24 to 26.5 °C and the ORP measurements started at –15 mV at reactor start up and ranged from –29 mV to –39 mV at steady state. More negative ORP values were generally associated with the shorter HRTs and less negative values were associated with longer HRTs. Runs using acetate addition observed slightly more negative ORP values. Analysis of the samples took place on August 18-19, 1998 and these data are in Appendix 2. A summary of important test site influent conditions and reactor conditions at 'steady state' are summarized in Tables 4.6 and 4.7, respectively.

Table 4.6 Snoqualmie Falls Site Test Reactor HRT and Influent Substrate Concentrations

Run #	HRT (min)	TBSCOD* (mg/L)	RBCOD* (mg/L)	NO_3-N (mg/L)	Acetate Addition
1	30.2			14.5	N
2	20.7			14.0	N
3	14.3			13.0	N
4	29.5			17.0	Y
5	21.4			16.5	Y
6	14.1			17.7	Y

*Influent CODs for runs 1 to 6 were not reliable due to sample storage issues and are thus not reported.

Table 4.7 Snoqualmie Falls Site Test Reactor Conditions and Reactor Substrate Concentrations

Run #	MLVSS (mg/L)	TEMP (°C)	pH	TBSCOD (mg/L)	RBCOD (mg/L)	NO_3-N (mg/L)
1	1487	24.0	7.32	N.D.	N.D.	16.5
2	1608	25.0	7.23	1.9	N.D.	10.6
3	1601	25.5	7.13	7.3	0.9	13.0
4	1739	26.0	7.19	23.6	14.9	5.1
5	1794	26.0	7.21	32.8	22.9	16.5
6	1702	26.5	7.30	39.3	23.4	11.7

The observed SDNRs versus HRT for all site reactor test runs conducted at Snoqualmie Falls are shown in Figure 4.13.

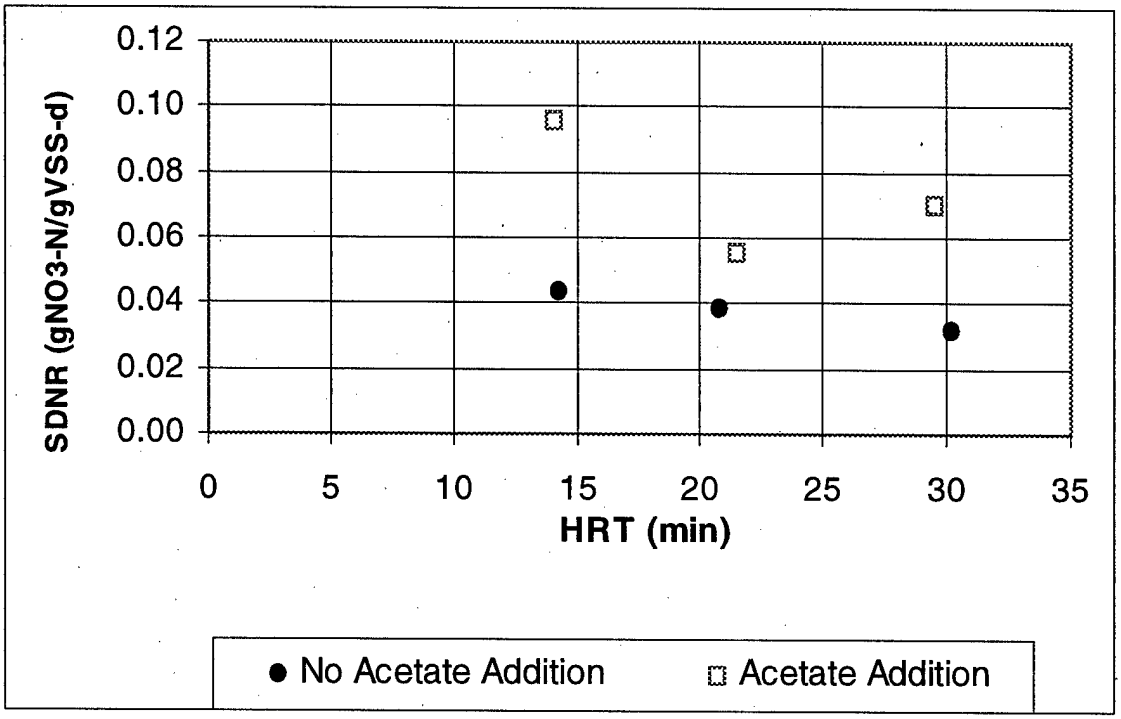

Figure 4.13 Observed SDNR (g NO_3-N/g VSS-d) versus HRT (min) for all Snoqualmie Falls Test Runs

The SDNRs vary from 0.03 to 0.1 mg NO_3-N/mg VSS-d and are lower than the rates observed at Olympus Terrace. A general decrease in SDNR is observed with increasing HRT but a good statistical fit to this trend is not likely.

As before, the SDNR was corrected for temperature and mixed liquor active biomass fraction and is plotted in terms of reactor TBSCOD and RBCOD concentrations in Figures 4.14 and 4.15, respectively.

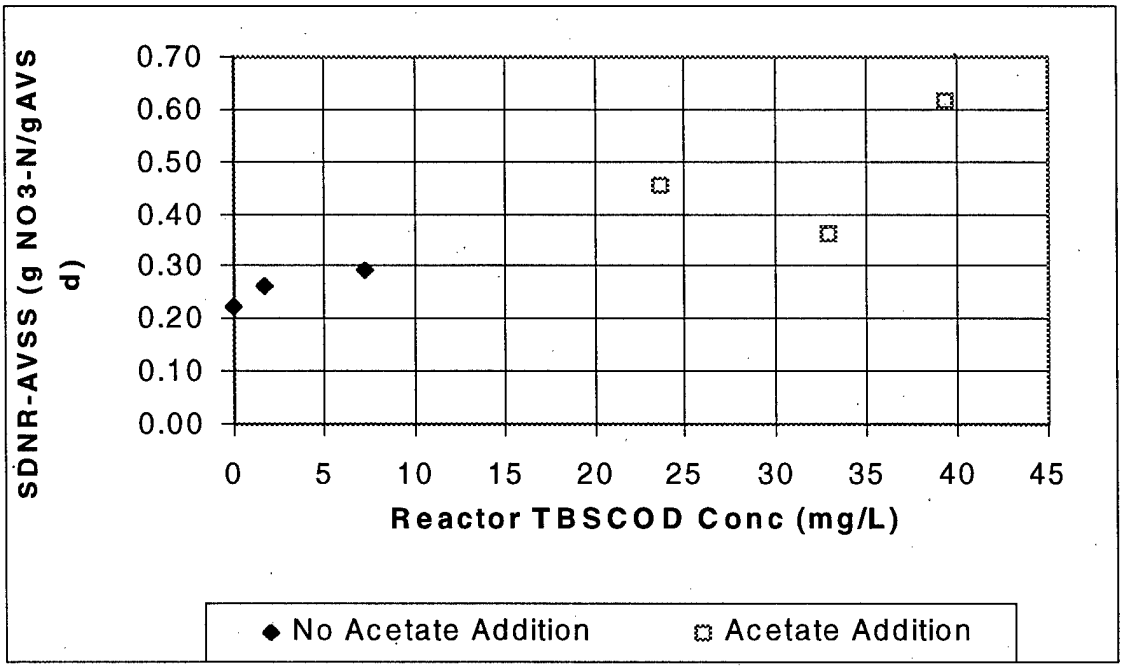

Figure 4.14 $SDNR_{AVSS}$ (g NO3-N/gAVSS-d) versus Reactor TBSCOD Concentration (mg/L) after Correcting Raw Data for Temperature and Active Mass Fraction for all Snoqualmie Falls Test Runs

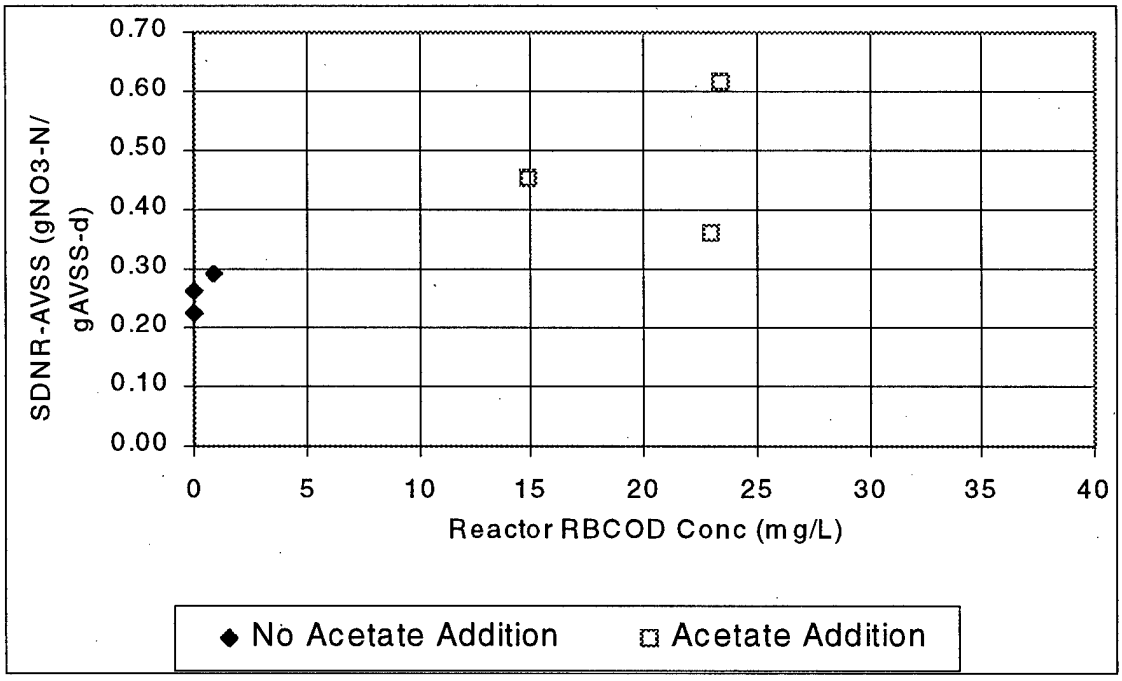

Figure 4.15 $SDNR_{AVSS}$ (g NO3-N/gAVSS-d) versus reactor RBCOD Concentration (mg/L) after Correcting Raw Data for Temperature and Active Mass Fraction for all Snoqualmie Falls Test Runs

The SDNR increases linearly with increased reactor TBSCOD and RBCOD concentrations with or without acetate addition. A maximum SDNR was not reached as was observed in the Olympus Terrace results and this was due to the fact that the reactor soluble COD concentration did not reach concentrations as high as was observed during the Olympus Terrace tests. Since similar HRT values were used for both site tests, this shows that the influent wastewater soluble COD was much lower for Snoqualmie Falls. The calculations to adjust the SDNR for active biomass is described next.

4.2.2 Adjustment of SDNR for Estimated Active Biomass

The same method used previously to estimate the active mass fraction was also used for the Snoqualmie Falls tests. The plant data is shown in Table 4.7 for the month of August:

Table 4.8 Snoqualmie Falls WWTP Average Operating Data for August 1998

Parameter	*Value*
SRT, days	23
Average Flow, MGD	0.292
Influent BOD, mg/L	76
Effluent BOD, mg/L	1.5
Solids Wasted per Day, lb TSS/day	285

A comprehensive listing of Snoqualmie Falls' general operating information is available at Appendix 2.

The net estimated active biomass yield for Snoqualmie Falls is determined using Equation 4.11 as:

$$Y_{bio} = (0.6)/(1+(0.08)*(23)) = 0.21 \text{ g TSS/g BOD}$$

The amount of biomass wasted per day is estimated using Equation 4.12.

$$P_{XBio} = (0.21 \text{ g/g})*(76 - 1.5 \text{ mg/L})*(0.292 \text{ MGD})*8.34 = 39 \text{ lb TSS/d}$$

Since the plant wasted sludge at an average of 285 pounds per day in August, the active biomass fraction was estimated using Equation 4.10 as:

AVSS = (39 lb biomass/d produced)/(284.7 lbs of sludge wasted/d) = 0.14

Thus, the estimated active biomass is 14% of the plant's MLSS concentration.

The plant reported a high degree of variability in solids wasting and SRT control over August so the estimated active biomass fraction based on the plant data was compared to the theoretical estimate. Snoqualmie Falls WWTP does not have primary treatment, so Y_{inerts} = 0.5 g TSS/g BOD is assumed. Using Equation 4.13, the active biomass was estimated as:

AVSS = 0.21 g TSS/g BOD / (0.21 g TSS/g BOD + 0.5 g TSS/g BOD) = 0.30

In order for the AVSS to be lower (0.14) by this calculation the inert solids yield would have to be much greater than 0.5 (1.21 g TSS/g BOD). Such a high inert yield would be unusual so the plant data used for the above calculation was considered inadequate. The plant was likely not at steady state and showed variable operating results the theoretical AVSS of 0.30 was used instead of the estimated AVSS based on the plant data.

4.2.3 Endogenous Respiration Batch Test Results

The mixed liquor endogenous uptake tests were conducted on August 18, 1998. After five hours of aeration 15 mg/L of nitrate was added to the mixed liquor and the Endogenous Oxygen Utilization Rate (OUR_{endog}) test was conducted as discussed in Chapter 2. The results of the OUR_{endog} test are shown in Figure 4.16 below:

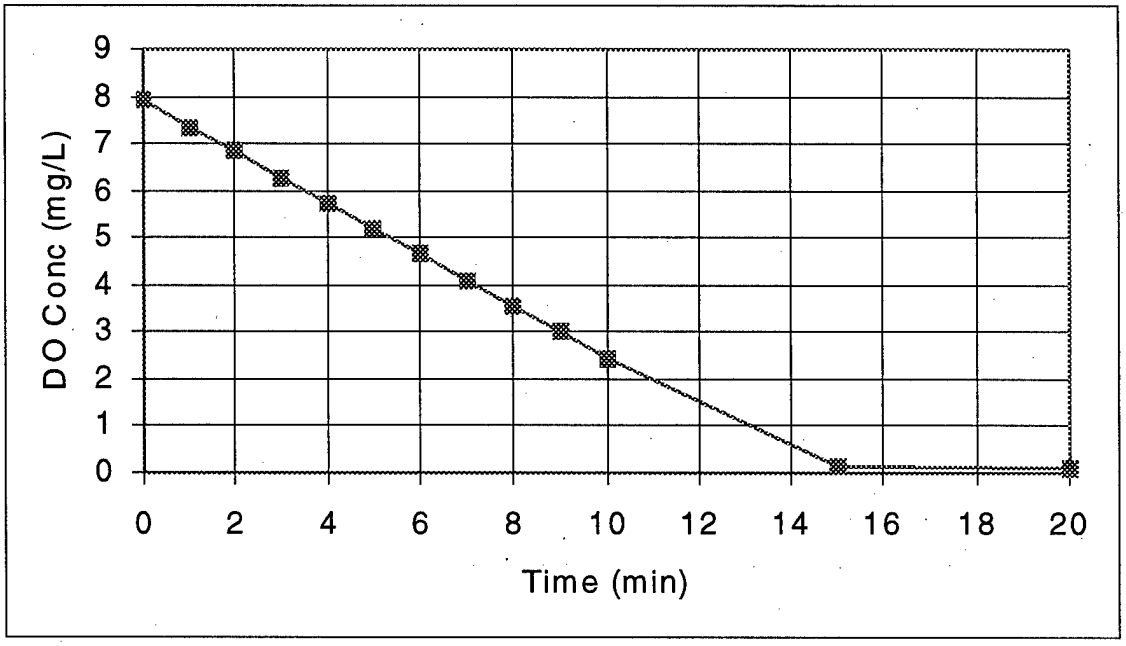

Figure 4.16 Snoqualmie Falls OUR_{endog} Test Results
(MLVSS = 4940 mg/L, Temp = 18.9 °C)

Using the Microsoft Excel SLOPE function the slope of the data over the linear oxygen utilization range was determined to be -0.54 mg O_2/L-min with an endogenous OUR of 32.5 mg O_2/L-hr. The Specific Endogenous Oxygen Utilization Rate (SEOUR) was determined by dividing the OUR by the MLVSS concentration of 4940 mg/L. The MLVSS concentration was based on analyses of two mixed liquor samples taken at the end of the OUR test. The resulting SEOUR for the test was 0.16 g O_2/gVSS-d. The temperature during the test was 18.9 °C. The SEOUR rate was corrected to 20 °C using Equation 2.3 to give an $SOUR_{20}$ of 0.167-g O_2/gVSS-d. Using the 1.42 g O_2 / g MLVSS equivalency ratio, the MLVSS decay coefficient was estimated as 0.12 g/g-d.

Next, the endogenous nitrate utilization rate (NUR_{endog}) was determined for the same mixed liquor as outlined in Chapter 2. The results of the NUR_{endog} test are shown in Figure 4.17 below:

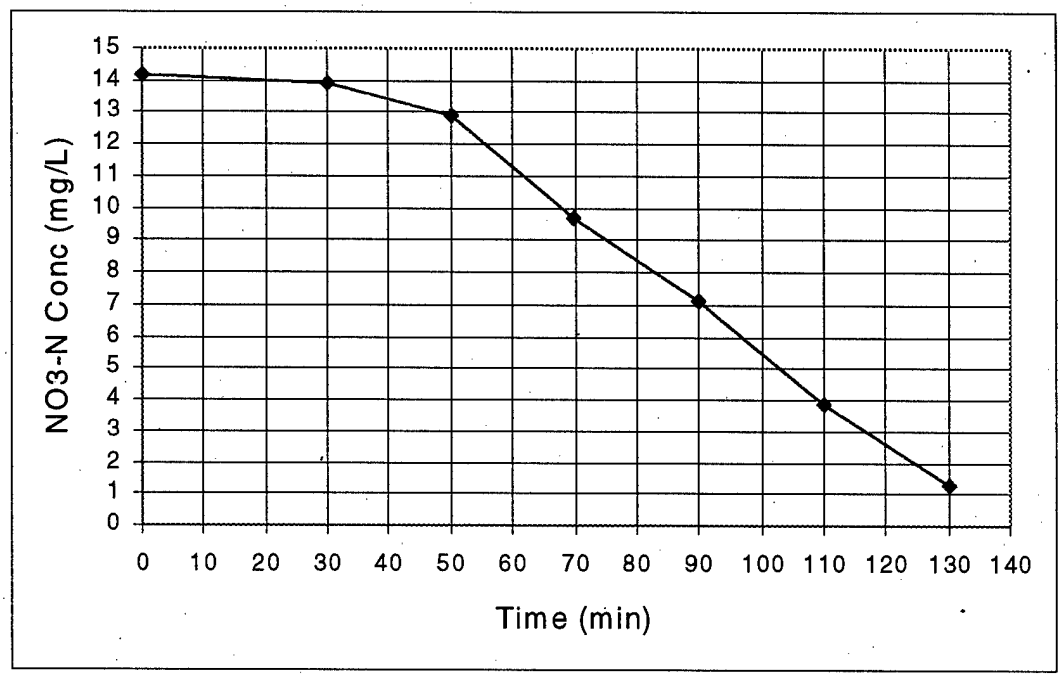

Figure 4.17 Snoqualmie Falls NUR_{endog} Test Results
(MLVSS = 4940, Temp = 24.1 °C)

A lag period of about 50-minutes occurred before a constant nitrate reduction rate is observed. This test response is similar to that observed for the Olympus Terrace results.

Using the Microsoft Excel SLOPE function, the slope of the data over the linear nitrate utilization range was determined as -0.145 mg NO_3-N/L-min or an endogenous NUR of 8.7 mg NO_3-N/L-hr. The resulting $SDNR_{endog}$ is 0.0437 g NO_3-N/gVSS-d. The temperature during the test was 24.1 °C. The $SDNR_{endog}$ was corrected to 20 °C resulting in an $SDNR_{endog20}$ of 0.0389 g NO_3-N/gVSS-d. This results in an equivalent specific oxygen uptake rate of (0.0389 g NO_3-N/gVSS-d)*(2.86 g O_2/g NO3-N) = 0.11 g O_2/gVSS-d.

The fraction of active biomass capable of nitrate reduction is thus estimated to be (0.11 g O_2/gVSS-d)/(0.167 g O_2/gVSS-d) or 0.66.

The SDNR based on the estimated active denitrifying biomass is shown in Figures 4.18 and 4.19 versus the reactor TBSCOD and RBCOD concentrations, respectively.

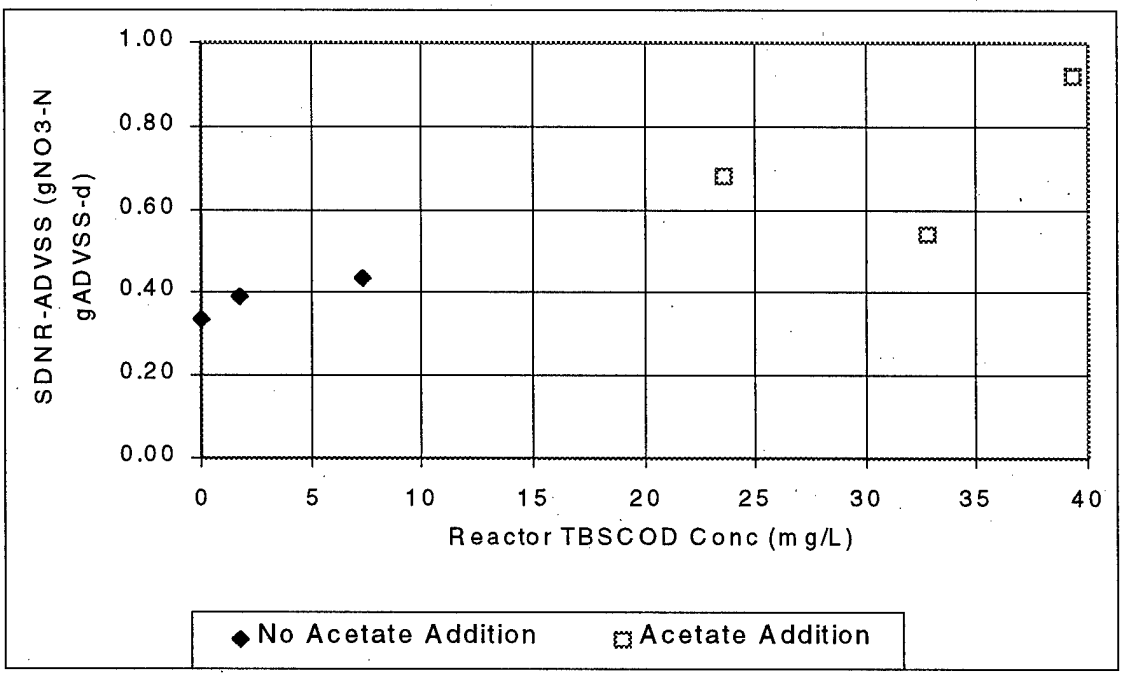

Figure 4.18 $SDNR_{ADVSS}$ (g NO3-N/gADVSS-d) versus Reactor TBSCOD Concentration (mg/L) for all Snoqualmie Falls Test Runs

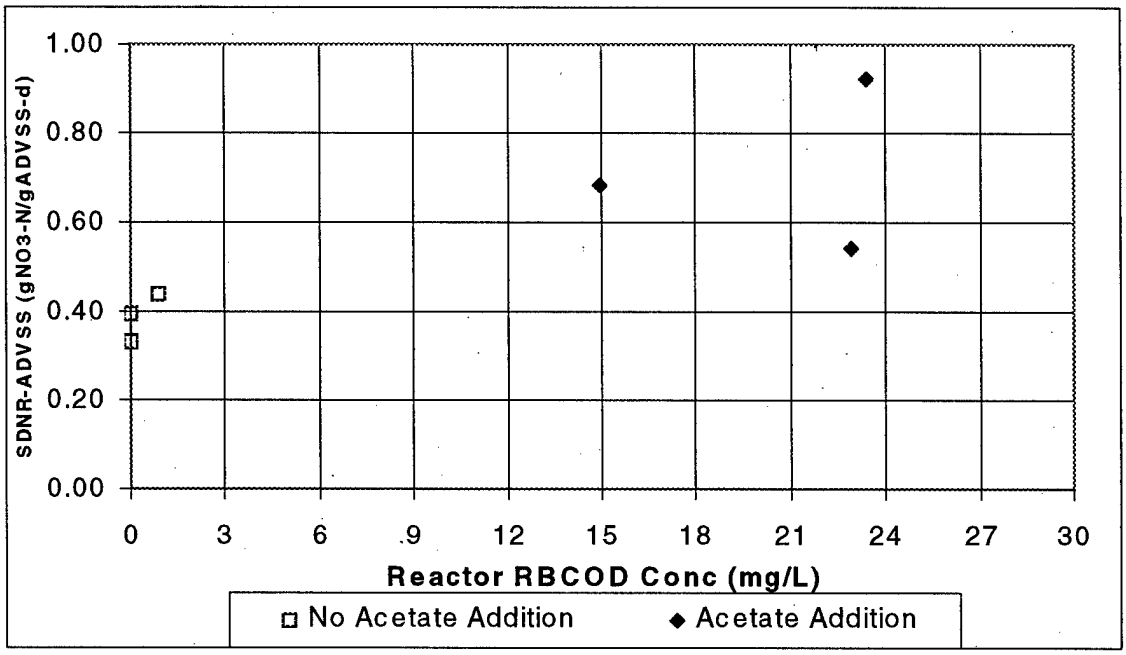

Figure 4.19 $SDNR_{ADVSS}$ (g NO3-N/gADVSS-d) versus Reactor RBCOD Concentration (mg/L) for all Snoqualmie Falls Test Runs

The corrected SDNRs are again directly proportional to the reactor TBSCOD and RBCOD concentrations. The $SDNR_{ADVSS}$ varied from 0.145 to 0.400 g NO_3/g ADVSS-d and a maximum SDNR is not indicated. These SDNRs represent the denitrification kinetics for actual nitrate respiring bacteria and are thus expected to be comparable to other WWTP denitrification kinetics.

4.2.4 COD and Nitrate Utilization

Due to inadequate influent COD sampling procedures used at Snoqualmie Falls, reliable COD utilization relationships could not be developed and analyzed.

4.3 Chamber's Creek Experimental Results

4.3.1 Results of Anoxic Reactor Experiments Conducted at Chamber's Creek

Initial experiments were conducted at Chamber's Creek WWTP on August 20, 1998. The site test anoxic reactor discussed in Chapter 3 was setup and operated. The raw data from the six experimental runs conducted is located at Appendix 3. The reactor temperatures for these tests ranged from 24.6 to 27 °C. These reactor temperatures were the highest of all runs conducted at any of the WWTPs. ORP measurements started at –5 mV at reactor start up and ranged from –29.4 mV to –39 mV at steady state. More negative ORP values were generally associated with the shorter HRTs and higher ORP values were associated with longer HRTs. HRT was controlled by reactor flow rates from the wastewater and RAS feed sources. Runs with acetate addition generally had ORP readings from 0 to 5mV lower than runs without acetate addition. Relative ORP values between the runs with and without acetate were similar for similar HRT time periods, however the reactor temperature for acetate runs averaged two degrees cooler than runs conducted earlier in the day without acetate. Analytical analysis of the first reactor samples took place on August 21-22, 1998. The results of the analytical analysis are in Appendix 3.

On September 24, 1998 a second set of experiments were conducted at the Chamber's Creek WWTP. The objectives of these additional experiments was to obtain denitrification kinetic information for the anoxic site test reactor at lower reactor substrate concentrations and to simulate plants with longer anoxic stage detention times. The anoxic site reactor temperatures were much lower during these experiments, ranging from 18 to 19 °C. Reactor ORP measurements however were much more negative than during 1st set of experiments, ranging from –52.3 mV to –68.9 mV. ORP reading tended to be lower (more negative) at the longer HRTs.

The results of the analytical analysis for all experimental runs conducted at Chamber's Creek are located at Appendix 3. A summary of important test site influent conditions and

reactor conditions at 'steady state' sampling that applied for each experimental run is shown in Tables 4.9 and 4.10, respectively.

Table 4.9 Chamber's Creek Site Test HRT and Influent Substrate Concentrations

Run #	HRT (min)	TBSCOD* (mg/L)	RBCOD* (mg/L)	NO_3-N (mg/L)	Acetate Addition
1	30.2			15.3	N
2	20.7			14.4	N
3	14.7			13.7	N
4	29.5			15.3	Y
5	21.4			14.9	Y
6	14.1			15.0	Y
7	62.5	26.7	14.8	21.4	N
8	62.5	65.7	48.5	22.4	Y
9	14.4	78.5	63.9	20.6	Y

*Influent CODs for runs 1 to 6 were not reliable due to sample storage issues and are thus not reported.

Table 4.10 Chamber's Creek Site Test Reactor Conditions and Reactor Substrate Concentrations

Run #	MLVSS (mg/L)	TEMP (°C)	pH	TBSCOD (mg/L)	RBCOD (mg/L)	NO_3-N (mg/L)
1	1257	27.0	7.32	25.2	5.9	13.1
2	1361	26.8	7.23	53.6	10.6	13.1
3	1471	27.0	7.13	55.5	16.1	12.7
4	1338	25.2	7.19	42.6	19.3	11.8
5	1382	25.0	7.21	63.6	30.0	12.2
6	1310	24.6	7.30	66.7	35.6	13.7
7	1673	18.0	7.82	26.7	14.8	16.8
8	1800	18.8	8.08	24.5	25.8	13.1
9	1827	19.0	7.75	62.0	50.2	18.6

A complete set of the raw data is available at Appendix 3.

SNDRs were determined using Equation 4.3. The observed SDNRs versus HRT for all site reactor test runs conducted at Chamber's Creek are shown in Figure 4.20.

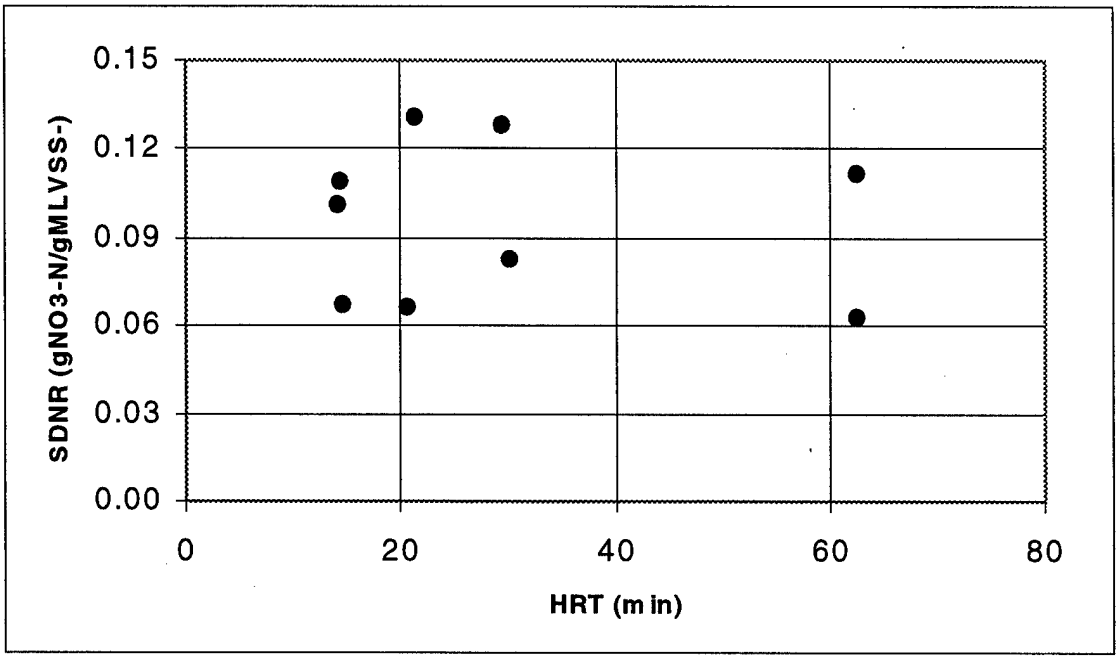

Figure 4.20 Observed SDNR (g NO_3-N/g VSS-d) versus HRT (min) for all Chamber's Creek Test Runs

The SNDRs vary from 0.066 to 0.131 g NO_3-N/g VSS-d and are higher than the rate observed at Snoqualmie Falls but lower than the rates observed at Chamber's Creek. As with the other plants, a good statistical fit is unlikely.

The data was examined to see how well the observed SDNRs fit the SDNR versus F/M Ratio relationship by Burdick et al. (1982). The observed SDNR versus F/M ratios for experimental runs 7 to 9 is shown in Figure 4.21 (runs 1-6 were not included due to lack of reliable influent COD values).

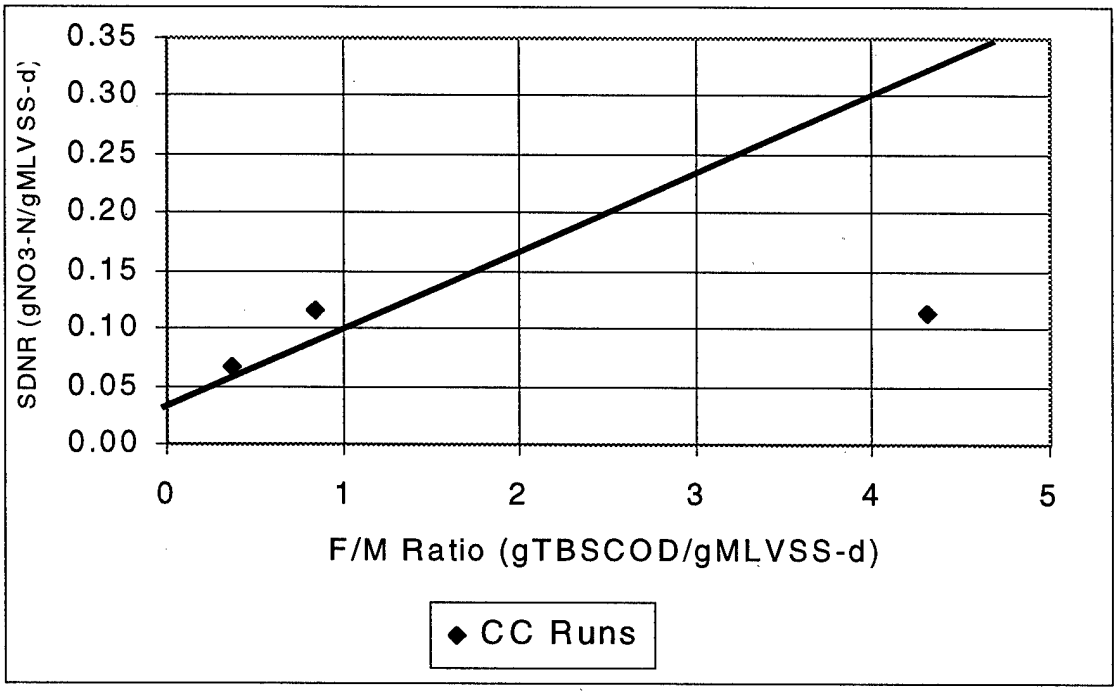

Figure 4.21 Observed SDNR (g NO3-N/gVSS-d) versus F/M Ratio (gTBSCOD/gVSS-d) for Site Reactor Test Runs 7 to 9 Conducted at Chamber's Creek

The F/M EQN line represents the relationship between F/M and SDNR as proposed by Burdick et al. (1982) and calculated using Equation 4.4. The F/M equation is represented by the solid line in Figure 4.21. Observed SDNRs are close to that predicted by the equation at F/M ratios less than 1.0, but the data point at the higher F/M is well below the predicted SDNR value.

As before the SNDR was corrected for temperature and the mixed liquor active biomass fraction, and is plotted versus reactor TBSCOD and RBCOD concentrations in Figures 4.22 and 4.23, respectively.

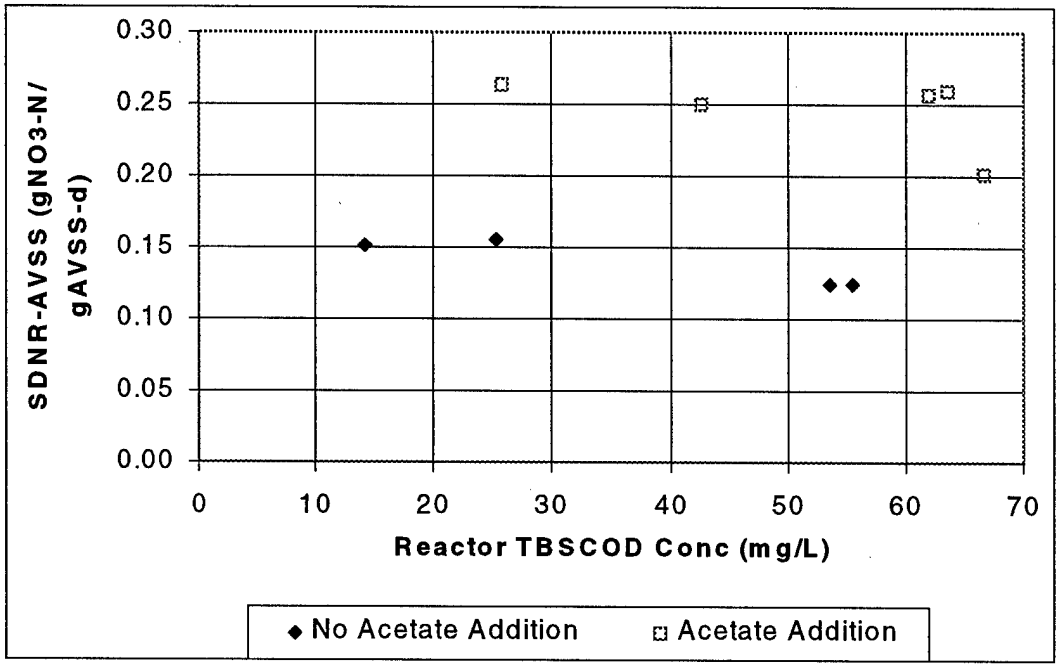

Figure 4.22 $SDNR_{AVSS}$ (g NO3-N/gAVSS-d) versus reactor TBSCOD Concentration (mg/L) after Correcting Raw Data for Temperature and Active Mass Fraction for all Chambers Creek Test Runs

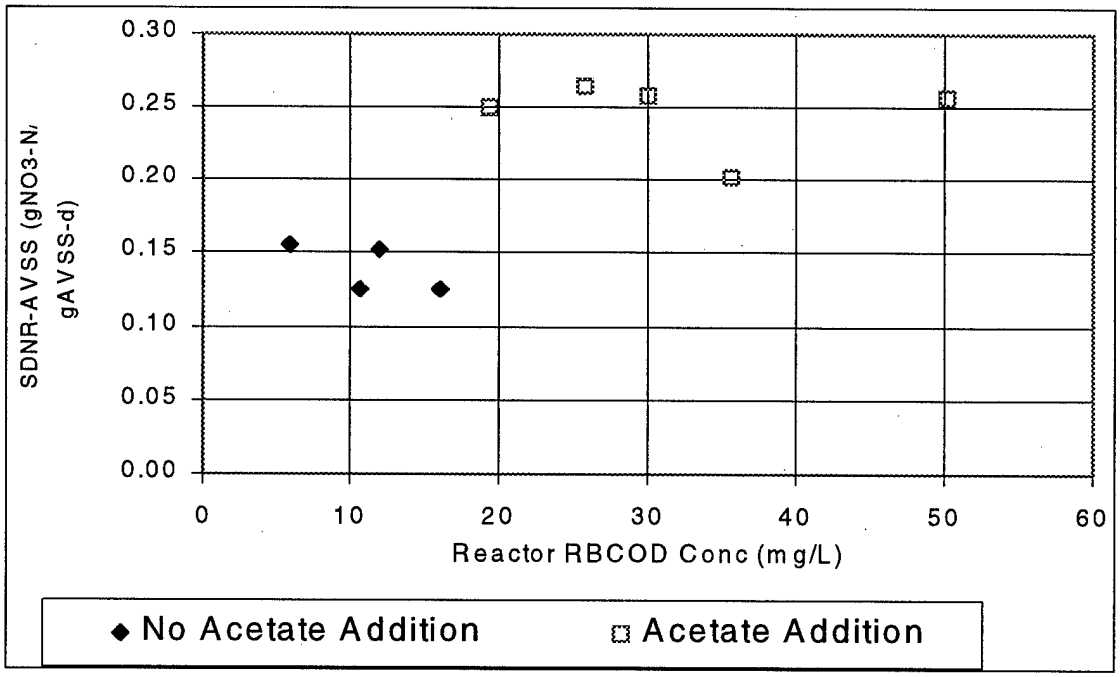

Figure 4.23 $SDNR_{AVSS}$ (g NO3-N/gAVSS-d) versus reactor RBCOD Concentration (mg/L) after Correcting Raw Data for Temperature and Active Mass Fraction for all Chamber's Creek Test Runs

The SDNR increases linearly with increased reactor TBSCOD and RBCOD concentrations with or without acetate addition. A maximum SDNR was reached as was observed in the Olympus Terrace results. Though similar HRT values were used for all site tests, this steady state anoxic reactor TBSCOD and RBCOD are in a range between the observed for Olympic Terrace and Snoqualmie Falls. This shows that the influent wastewater soluble COD was lower than the SCOD concentrations observed at Olympus Terrace and higher than the SCOD reactor concentrations present at Snoqualmie Falls. The method to adjust the SDNR for active biomass is described next.

4.3.2 Adjustment of SDNR for Estimated Active Biomass

The same method used previously to estimate the active mass fraction was also used for the Snoqualmie Falls tests. The plant data is shown in Table 4.11 for the month of August:

Table 4.11 Chamber's Creek WWTP Operating Data for August 1998

Reported Value	*Value*
SRT, days	3.6
Average Flow, MGD	13.61
Influent BOD, mg/L	106
Effluent BOD, mg/L	3.3
Solids Wasted per Day, lb TSS/day	12433

A comprehensive listing of Chamber's Creek's general operating information is available at Appendix 3.

The net active estimated biomass yield for Chamber's Creek is determined using Equation 4.11:

$$Y_{bio} = (0.6)/(1+(0.08)*(3.6)) = 0.47 \text{ g TSS/g BOD}$$

The amount of biomass wasted per day in August was determined using Equation 4.12.

$$P_{XBio} = (0.47\ g/g)*(106 - 3.3\ mg/L)*(13.61\ MGD)*8.34 = 5479\ lb\ TSS/d$$

Since the plant wasted sludge at an average of 12433 lb pounds TSS per day in August, the active biomass fraction was estimated using Equation 4.10 as:

$$AVSS = (5479)/(12433) = 0.44$$

Thus, the estimated active biomass is 44% of the plant's MLVSS concentration.

As a check on the reasonableness of this estimation, the theoretical active biomass was determined. Chamber's Creek WWTP does have primary treatment, so $Y_{inerts} = 0.3$ g TSS/g BOD is assumed. Using Equation 4.13, the active biomass was estimated as:

$$AVSS = 0.47\ g\ TSS/g\ BOD\ /\ (0.47\ g\ TSS/g\ BOD + 0.3\ g\ TSS/g\ BOD) = 0.61$$

The estimated AVSS using the plant data is 72% of the theoretical AVSS of 0.61 that assumes growth yields and inert solids yields. Since the Chamber's Creek plant had an estimated steady state operation and showed careful procedures for determining wasting, mixed liquor concentrations and all other important parameters the AVSS of 0.44 based on the plant data was used to adjust the SDNRs.

4.3.3 Endogenous Respiration Batch Test Results

The mixed liquor endogenous uptake tests were conducted on August 21, 1998. After four hours of aeration approximately 10 mg/L of nitrate was added to the mixed liquor and the Endogenous Oxygen Utilization Rate (OUR_{endog}) test was conducted as discussed in Chapter 3. The results of the OUR_{endog} test are shown in Figure 4.24.

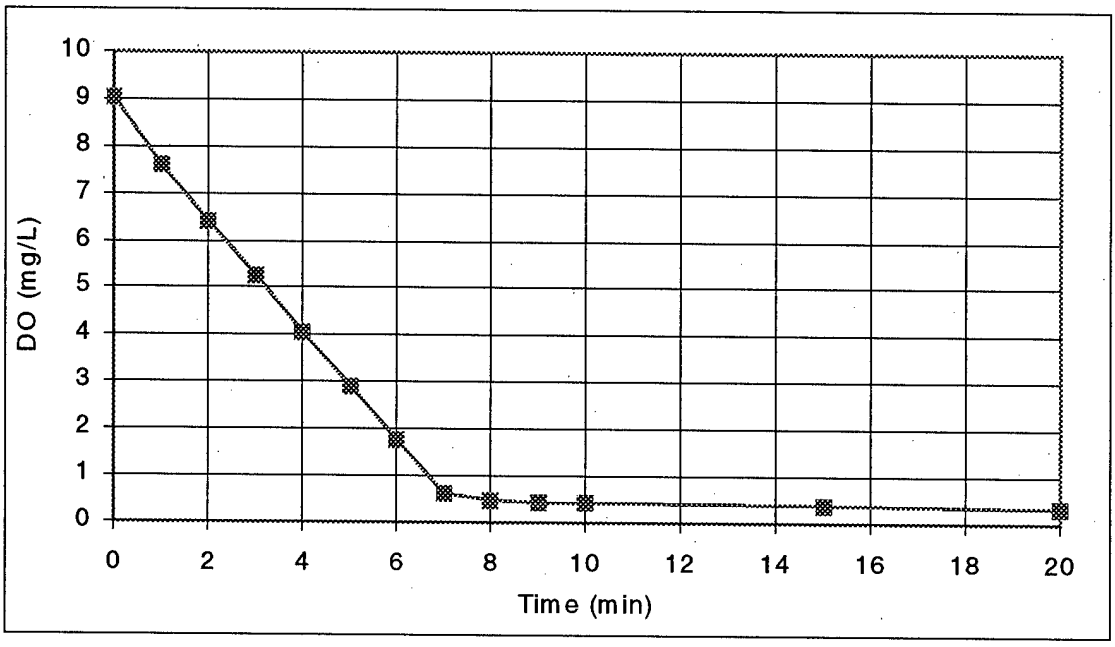

Figure 4.24 Chamber's Creek OUR_{endog} Test Results
(MLVSS = 5310 mg/L, Temp = 17.8 °C)

Using the Microsoft Excel SLOPE function, the slope of the data over the linear oxygen utilization range was determined as -1.17 mg O_2/L-min with an endogenous OUR of 70.4 mg O_2/L-hr. The Specific Endogenous Oxygen Utilization Rate (SOUR) was determined by dividing the OUR by the MLVSS concentration of 5310 mg/L. The MLVSS concentration was based the analysis of two mixed liquor samples taken at the end of the OUR test. The resulting SEOUR for the test was 0.318 g O_2/gVSS-d. The temperature during the test was 17.8 °C. The SEOUR rate was corrected to 20 °C using Equation 2.3 to give an $SEOUR_{20}$ of 0.339-g O_2/gVSS-d. Using the 1.42 g O_2 / g MLVSS equivalency ratio, the MLVSS decay coefficient was estimated as 0.24 g/g-d. As in the Olympus Terrace results, the observed decay coefficient was much higher than values typically reported in literature.

Next, the endogenous nitrate utilization rate (NUR_{endog}) was determined for the same mixed liquor as outlined in Chapter 2. The results of the NUR_{endog} test are shown in Figure 4.25.

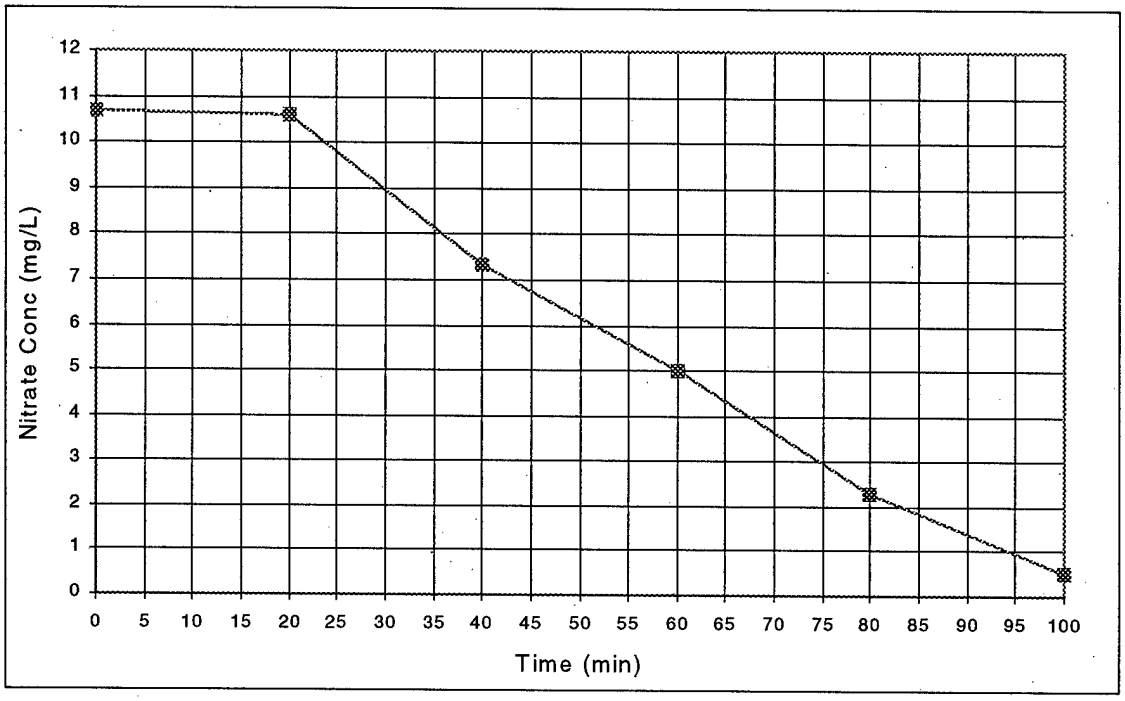

Figure 4.25 Chamber's Creek NUR_{endog} Test Results
(MLVSS = 5310, Temp = 21.6 °C)

A 20-minute lag time was noted before any appreciable nitrate removal took place. This lag time was shorter that the lag times observed in the other NUR_{endog} tests. However, this sample was allowed extra time to purge excess DO after the OUR test was conducted and before the NUR test commenced. This observation supported the acclimation findings previously noted.

Using the Microsoft Excel SLOPE function, the slope of the data over the linear nitrate utilization range was determined as -0.125 mg NO_3-N/L-min with an endogenous NUR of 7.5 mg NO_3-N/L-hr. The $SDNR_{endog}$ was then calculated to be 0.0375 g NO_3-N/gVSS-d. The temperature during the test was 21.6 °C. The $SDNR_{endog}$ was corrected to 20 °C using Equation 2.3, to give an $SDNR_{endog20}$ of 0.0358 g NO_3-N/gVSS-d. This results in an equivalent specific oxygen uptake rate of (0.0358 g NO_3-N/gVSS-d)*(2.86 g O_2/g NO3-N) = 0.10 g O_2/gVSS-d. The fraction of active biomass capable of nitrate reduction is thus estimated to be (0.1 g O_2/gVSS-d)/(0.339 g O_2/gVSS-d) or 0.295.

The SDNR site test results was then further modified to express the SDNR relative to the biomass that could use nitrate by dividing the $SDNR_{AVSS}$ by the fraction of denitrifying biomass. The resulting SNDR represents the nitrate removal kinetics by the estimated active biomass capable of denitrification. The $SDNR_{ADVSS}$ versus the reactor TBSCOD and RBCOD concentrations are shown in Figures 4.26 & 4.27, respectively.

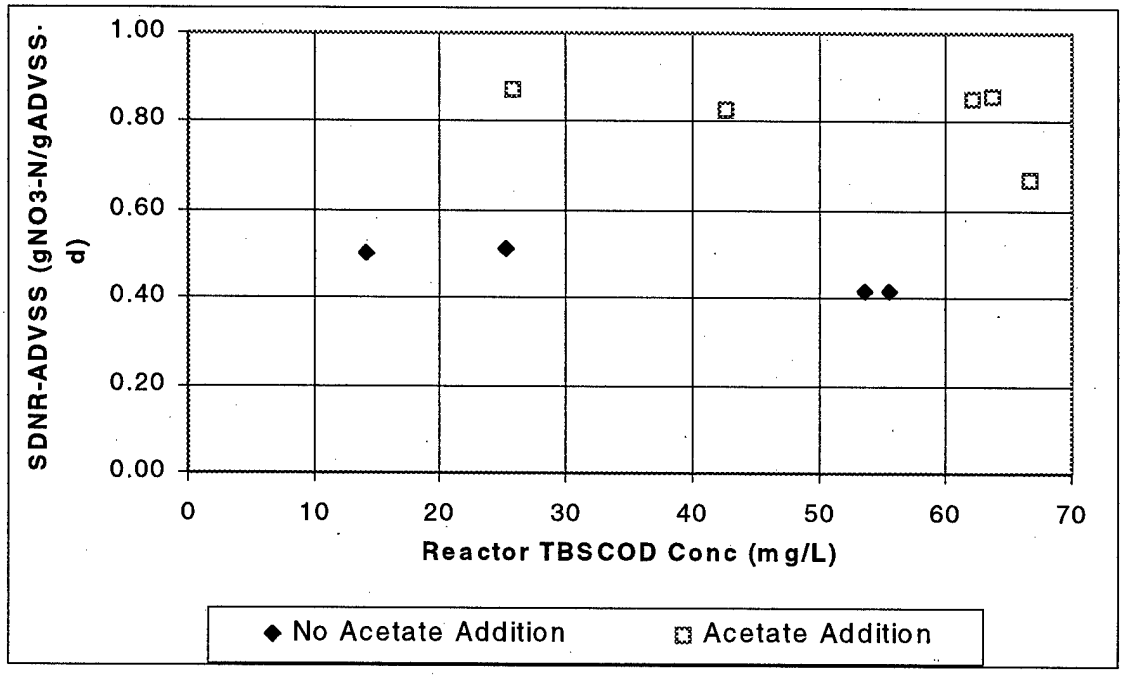

Figure 4.26 $SDNR_{ADVSS}$ (g NO3-N/gADVSS-d) versus Reactor TBSCOD Concentration (mg/L) for all Chamber's Creek Test Runs

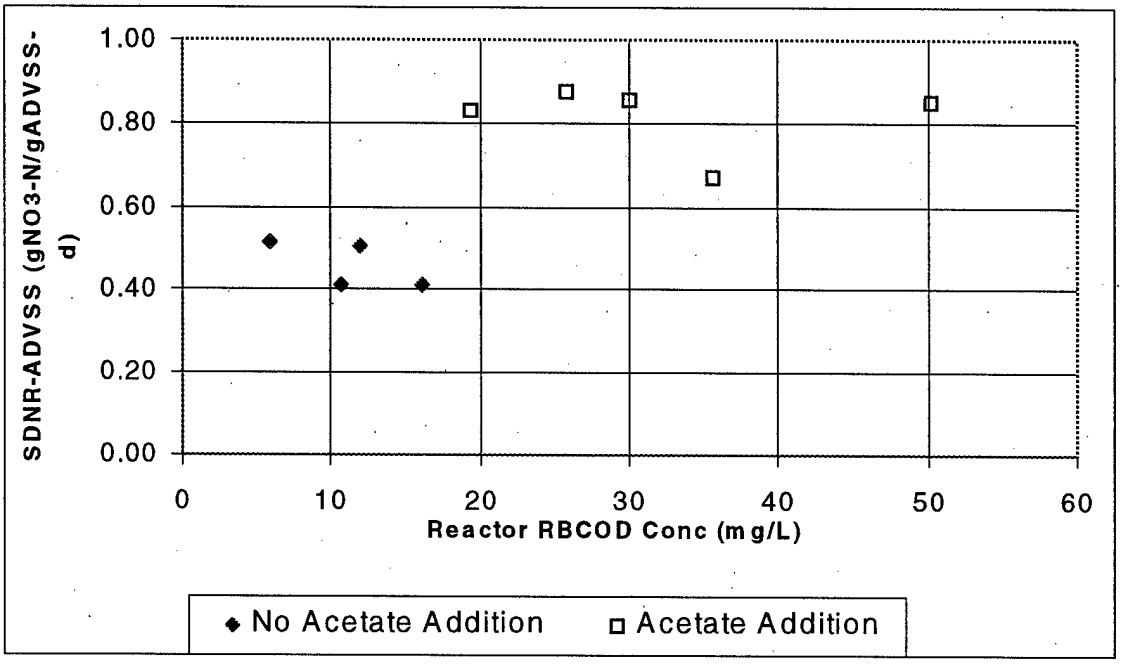

Figure 4.27 $SDNR_{ADVSS}$ (g NO3-N/gADVSS-d) versus Reactor RBCOD Concentration (mg/L) for all Chamber's Creek Test Runs

After the active denitrifying biomass fraction correction, the SNDRs continued to generally increase with both RBCOD and BSCOD reactor concentrations. The $SDNR_{ADVSS}$ varied from 0.414 to 0.874 g NO_3/g ADVSS-d. A maximum SDNR of about 0.874 g NO_3/g ADVSS-d was observed.

4.3.4 COD and Nitrate Utilization

The substrate utilization rates of biomass during anoxic site tests conducted on September 24, 1998 are reported in Table 4.12.

Table 4.12 Chamber's Creek Specific COD Utilization Rates for Test Runs 7-10 (Based on Active Biomass)

Run	HRT	Reactor Soluble COD Concentrations (mg/L)			Specific COD Utilization Rates (gCOD/gXa-d)			Acetate
	(min)	TBSCOD	BSCOD	RBCOD	R_{TBSCOD}	R_{BSCOD}	R_{RBCOD}	Addition
7	62.5	5.1	2.2	2.9	1.31	1.01	0.3	N
8	62.5	22.7	ND	22.7	3.97	1.78	2.19	Y
9	14.4	25.5	11.8	13.7	6.79	1.15	5.64	Y

Unreliable quantification of the influent COD fraction concentrations during runs 1 to 6 at Chamber's Creek resulted in their exclusion from the COD utilization analysis.

This table shows that both BSCOD and RBCOD are consumed simultaneously in the reactor. The observed specific COD utilization rates do correlate with the reactor soluble COD concentrations, indicating higher specific COD utilization rates with higher reactor soluble COD concentrations for both RBCOD and TBSCOD concentrations. The SDNR results in Figure 4.27 showed that higher SDNRs occurred with higher reactor RBCOD concentration and that acetate addition increased the SDNR, especially when added to the zones experiencing lower initial reactor concentrations. So in this case a similar trend is seen with higher SDNRs and higher specific COD utilization rates with reactor soluble COD concentrations as expected from bio-kinetic models. The COD and nitrate utilization rates are compared in Table 4.13 to determine if a direct relation exists. Again an approximate constant ratio of nitrate and COD utilization would be expected if COD was consumed with little or no cellular storage.

Table 4.13 shows that nitrate used for endogenous respiration and for RBCOD removal.

Table 4.13 Calculated COD/Nitrate Removal Consumption Ratios and Applied F/M Ratios based on reactor TBSCOD Concentration during Chamber's Creek Test Runs

RUN	HRT	MLSS	Endog Nitrate Usage	Observed SNDR	SDNR for	Spec TBSCOD	COD/ NO3	F/M Ratio
				(ADVSS)	RBCOD rem	Util		
	(min)	(mg/L)	(g/g-d)	(mg/L)	(g/g-d)	(g/gXa-d)	(g/g)	(g/g-d)
7	62.5	1673	0.0165	0.503	0.487	1.31	2.69	0.37
8	62.5	1800	0.0165	0.874	0.858	3.97	4.62	0.84
9	14.4	1826	0.0165	0.851	0.835	6.79	8.1	4.3

An estimated COD/NO3-N ratio was calculated from the estimated active biomass yield of 0.47 g VSS/g COD as discussed in section 4.1.5 resulting in an estimated ratio of 8.6 g COD/g NO3-N used. The COD/NO3-N consumption ratios for runs 7-9 vary to large degree and are all less than this theoretical value. Substrate storage is not expected in view of the lower rate as was indicated by the Olympus Terrace results. The lower ratio

suggests that the COD/NO3 ratio calculation is in error or error existed with the data collected. The discrepancy is most likely due related to the calculation method. The endogenous respiration SDNR was only estimated and the COD utilization was only based on soluble COD removal. Some COD would be supplied by the hydrolysis of influent biodegradable VSS. The wastewater was allowed to settle in the feed reservoir but some degradable VSS could have been supplied to the anoxic reactor. Thus the COD/NO_3-N consumption ratio can only be used as a rough estimate to indicate general trends in the results.

The F/M ratios shown indicate that the first two anoxic zones experienced low substrate loading conditions and run 9 experience a fairly high substrate loading condition. Specific COD utilization rates also appears to be directly related to the F/M Ratio as would be expected.

4.4 LOTT WWTP Experimental Results

4.4.1 Results of Anoxic Reactor Experiments Conducted at LOTT

Initial site field experiments were conducted at the LOTT WWTP on August 25,27 1998. The site test anoxic reactor discussed in Chapter 3 was setup and operated. The raw data from the six experimental runs conducted is located at Appendix 4. The reactor temperatures for these tests ranged from 20.0 to 24 °C. ORP measurements started at zero mV at reactor start up and ranged from –24.5 mV to –40 mV at steady state. More negative ORP values were generally associated with the shorter HRTs and higher ORP values were associated with longer HRTs. HRT was controlled by reactor flow rates from the wastewater and RAS feed sources. Runs with acetate addition generally had more negative ORP readings.

On September 22, 1998 a second set of experiments were conducted at the Chamber's Creek WWTP. The objectives of these additional experiments were to obtain denitrification kinetic information longer anoxic reactor HRTs and lower reactor soluble COD concentrations. The anoxic site reactor temperatures were generally lower during these experiments, ranging from 19.1 to 22 °C. Reactor ORP measurements however were much more negative than during 1st set of experiments, ranging from –37.7 mV to –66.3 mV. ORP reading tended to be lower (more negative) at the longer HRTs.

The raw data for all experiments conducted at LOTT are located at Appendix 4. A summary of important test site influent conditions and reactor conditions at steady state for experimental run is shown in Tables 4.14 and 4.15 respectively.

Table 4.14 LOTT Site Test Reactor HRT and Influent Substrate Concentration

Run #	HRT (min)	TBSCOD* (mg/L)	RBCOD* (mg/L)	NO_3-N (mg/L)	Acetate Addition
1	33.3			14.3	N
2	22.5			11.6	N
3	15.7			13.5	N
4	30.3			22.0	Y
5	21.9			20.8	Y
6	16.2			20.2	Y
7	59.3	49.9	36.0	23.1	N
8	14.7	45.5	30.3	22.9	N
9	63.9	76.9	55.3	24.7	Y
10	14.0	89.4	75.2	25.1	Y

*Influent CODs for runs 1 to 6 were not reliable due to sample storage issues and are thus not reported.

Table 4.15 LOTT Site Test Reactor Conditions and Reactor Substrate Concentrations

Run #	MLVSS (mg/L)	TEMP (°C)	PH	TBSCOD (mg/L)	RBCOD (mg/L)	NO_3-N (mg/L)
1	1125	20.8	7.27	47.3	18.0	7.2
2	1530	20.9	7.21	57.0	26.5	4.8
3	1238	21.1	7.13	64.9	45.0	9.5
4	1801	20.0	7.29	129.0	105.9	9.2
5	1894	21.8	7.26	140.3	110.0	10.1
6	1738	22.3	7.20	152.0	118.2	11.9
7	1227	19.1	7.85	9.8	11.0	14.2
8	1280	20.2	7.53	21.0	16.7	18.9
9	1453	21.1	8.08	25.5	33.5	3.5
10	1580	22.0	7.53	58.9	49.3	19.0

A complete set of the raw data is available at Appendix 4.

SNDRs were determined using Equation 4.3. The observed SDNRs versus HRT for all site reactor test runs conducted at LOTT are shown in Figure 4.28.

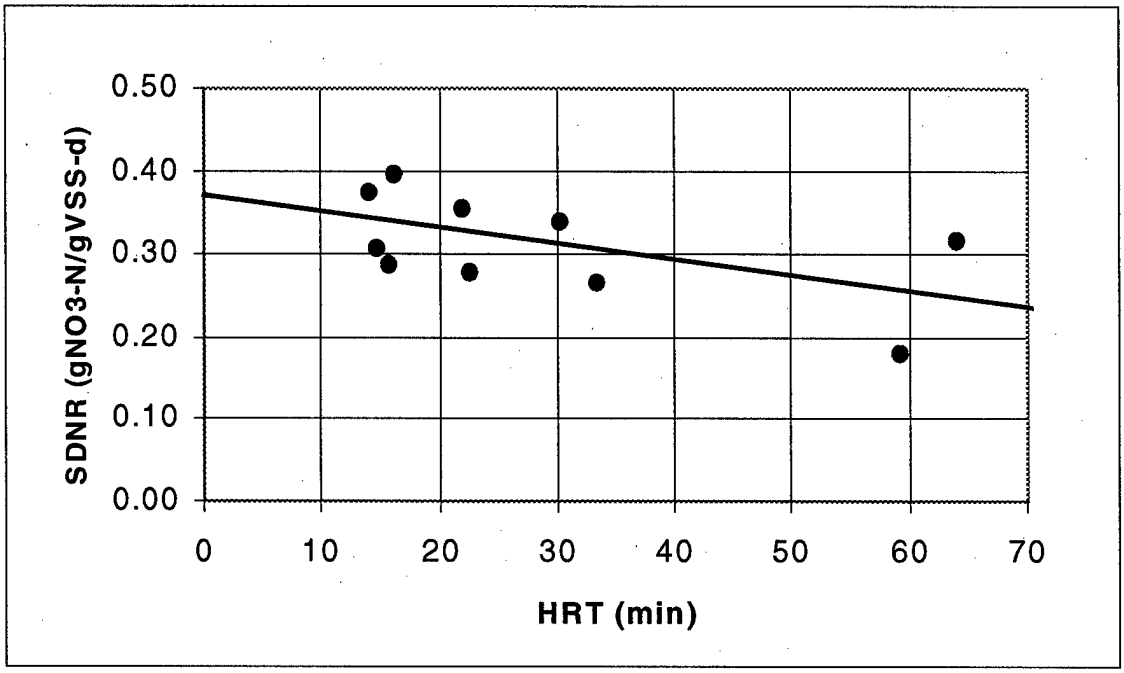

Figure 4.28 Observed SDNR (g NO_3-N/g VSS-d) versus HRT (min) for all Chamber's Creek Test Runs

The SNDRs vary from 0.176 to 0.424 mg NO_3-N/mg VSS-d and were the highest uncorrected SDNRs observed between all of the plants. A statistical fit of the data was made using Microsoft Excel slope and intercept function resulting in slope of –0.002 and an intercept of 0.38.

The observed SDNRs are compared to the predicted SDNRs using Equation 4.4, the modified form of the F/M Ratio relationship by Burdick et al. (1982). The observed SDNR versus F/M ratios for experimental runs 7 to 9 is shown in Figure 4.29 (runs 1-6 were not included due to lack of reliable influent COD values).

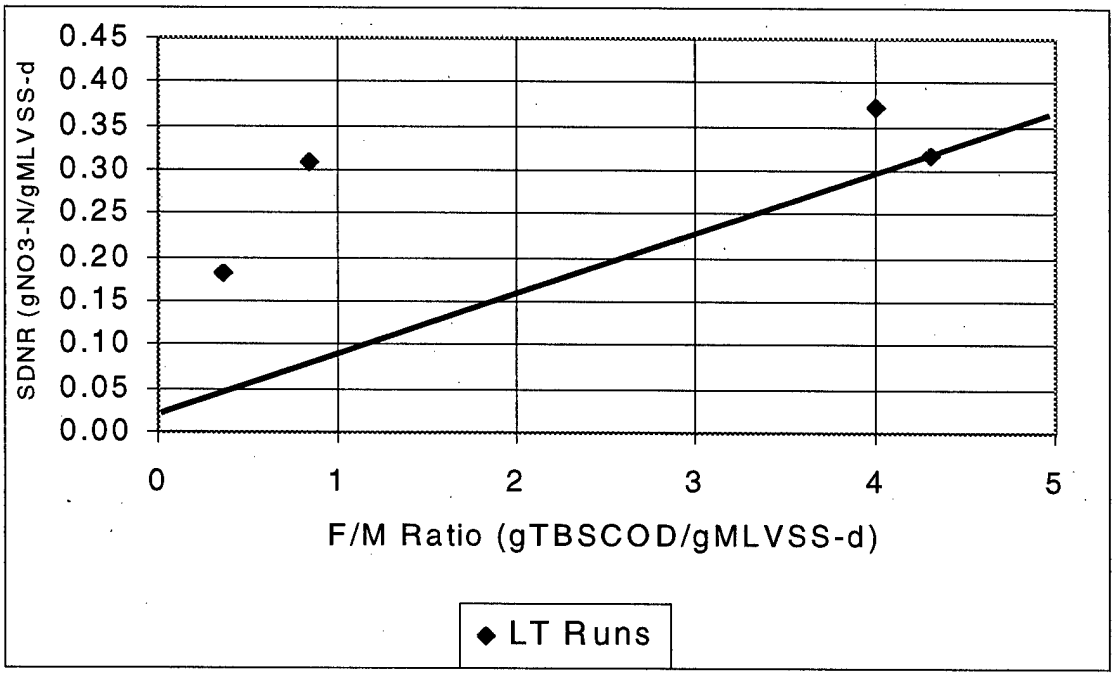

Figure 4.29 Observed SDNR (g NO3-N/gVSS-d) versus F/M Ratio (gCOD/gVSS-d) for Site Reactor Test Runs 7 to 10 Conducted at LOTT

The F/M equation is represented by the solid line in Figure 4.29. Observed SDNRs increased slightly with F/M Ratio for runs 7-10, but the values are higher than predicted by the F/M equation.

As before, the SDNR was corrected for temperature and mixed liquor active mass fraction and is plotted in terms of reactor TBSCOD and RBCOD concentrations in Figures 4.30 and 4.31 respectively.

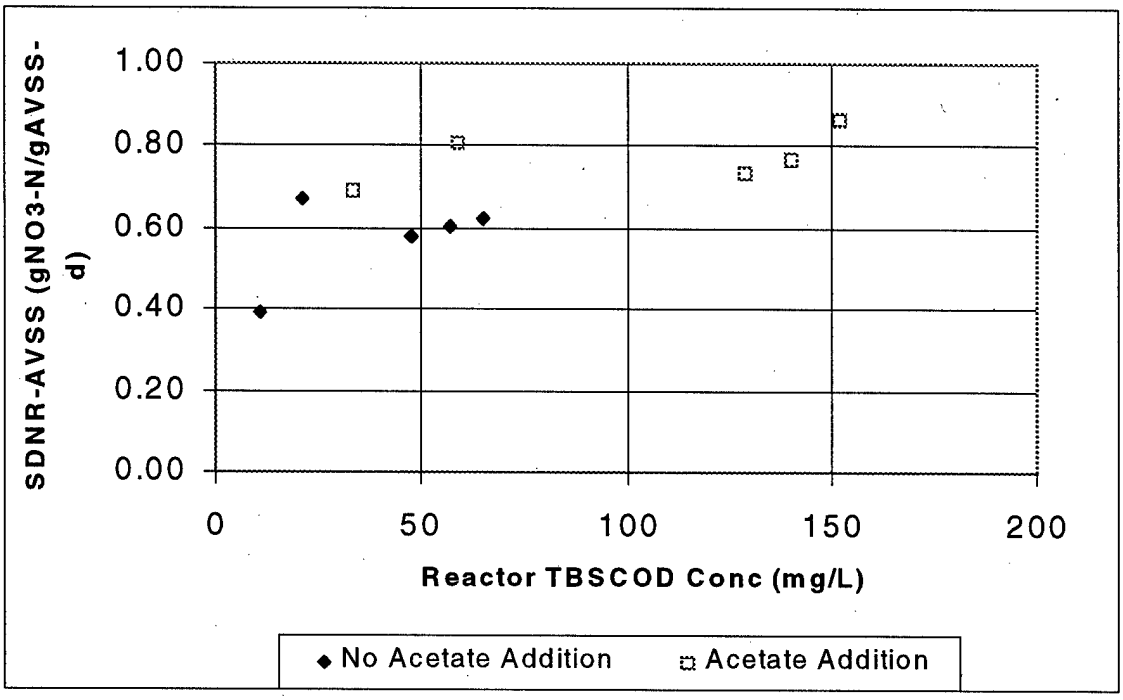

Figure 4.30 $SDNR_{AVSS}$ (g NO3-N/gAVSS-d) versus Reactor TBSCOD Concentration (mg/L) after Correcting Raw Data for Temperature and Active Mass Fraction for all LOTT Test Runs

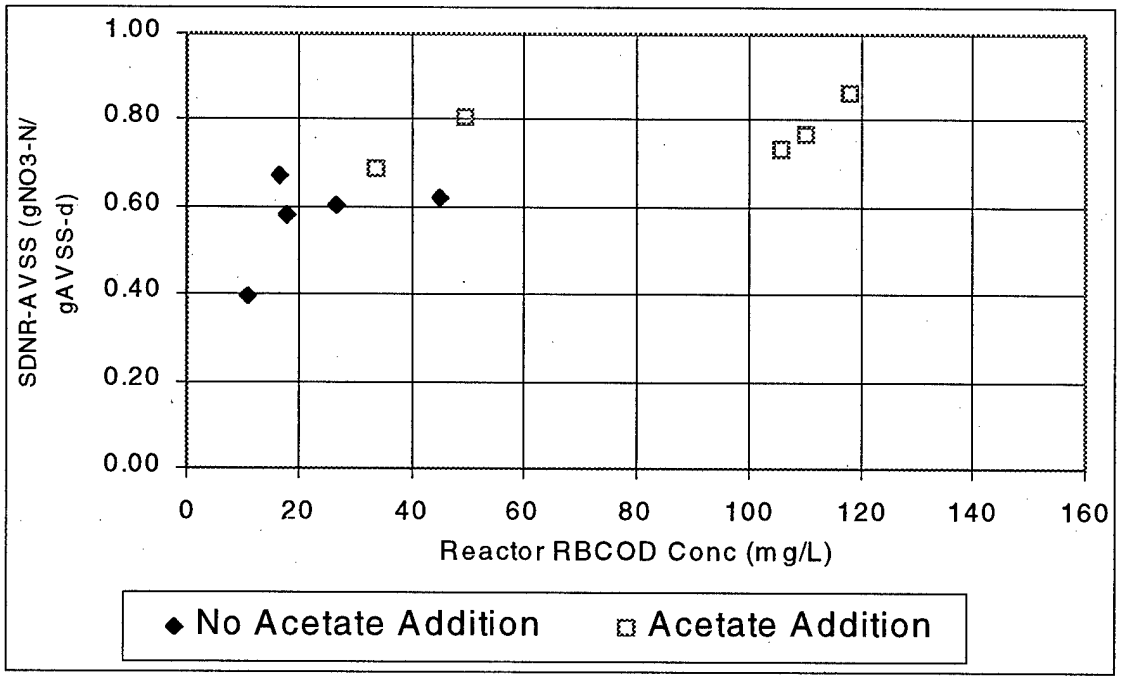

Figure 4.31 $SDNR_{AVSS}$ (g NO3-N/gAVSS-d)versus Reactor RBCOD Concentration (mg/L) after Correcting Raw Data for Temperature and Active Mass Fraction for all LOTT Test Runs

The SDNR increases linearly with increased reactor TBSCOD and RBCOD concentrations with or without acetate addition. A maximum SDNR was observed at higher soluble COD concentrations. The method to adjust the SDNR for active biomass is described next.

4.4.2 Adjustment of SDNR for Estimated Active Biomass

The same method used previously to estimate the active mass fraction was also used for the Snoqualmie Falls tests. The plant data is shown in Table 4.16 for the month of August.

Table 4.16 LOTT WWTP Operating Data for August 1998

Reported Value	*Value*
SRT, days	11.6
Average Flow, MGD	8.85
Influent BOD, mg/L	214
Effluent BOD, mg/L	3.3
Solids Wasted per Day, lb TSS/day	10495

A comprehensive listing of LOTT's general operating information is available at Appendix 4.

The net estimated active biomass yield for LOTT is determined using Equation 4.11 as:

$$Y_{bio} = (0.6)/(1+(0.08)*(11.6)) = 0.31 \text{ g TSS/g BOD}$$

The amount of biomass wasted per day is estimated using Equation 4.12.

$$P_{XBio} = (0.31 \text{ g/g})*(214 - 3.3 \text{ mg/L})*(8.85 \text{ MGD})*8.34 = 4820 \text{ lb TSS/d}$$

Since the plant wasted sludge at an average of 10495 pounds per day in August, the active biomass fraction was estimated using Equation 4.10 as:

$$AVSS = (4820)/(10495) = 0.46$$

Thus, the estimated active biomass is 46% of the plant's MLVSS concentration.

The theoretical active biomass fraction was determined to check reasonableness of estimated active biomass fraction. LOTT WWTP does have primary treatment, so Y_{inerts} = 0.3 g TSS/g BOD is assumed. Using Equation 4.13, the active biomass was estimated as:

AVSS = 0.31 g TSS/g BOD / (0.31 g TSS/g BOD + 0.3 g TSS/g BOD) = 0.51

Since the estimated AVSS of 0.46 is within 10% of the theoretical AVSS of 0.51, the estimated AVSS fraction based on the plant data is considered to be a reasonable estimate. This plant's collection and testing procedures for determining wasting, mixed liquor concentration and all other important parameters is extremely thorough and meticulous.

4.4.3 Endogenous Respiration Batch Test Results

The mixed liquor endogenous uptake tests were conducted on August 29, 1998. After four hours of aeration approximately 15 mg/L of nitrate was added to the mixed liquor and the Endogenous Oxygen Utilization Rate (OUR_{endog}) test was conducted as discussed in Chapter 2. The results of the OUR_{endog} test are shown in Figure 4.32 below:

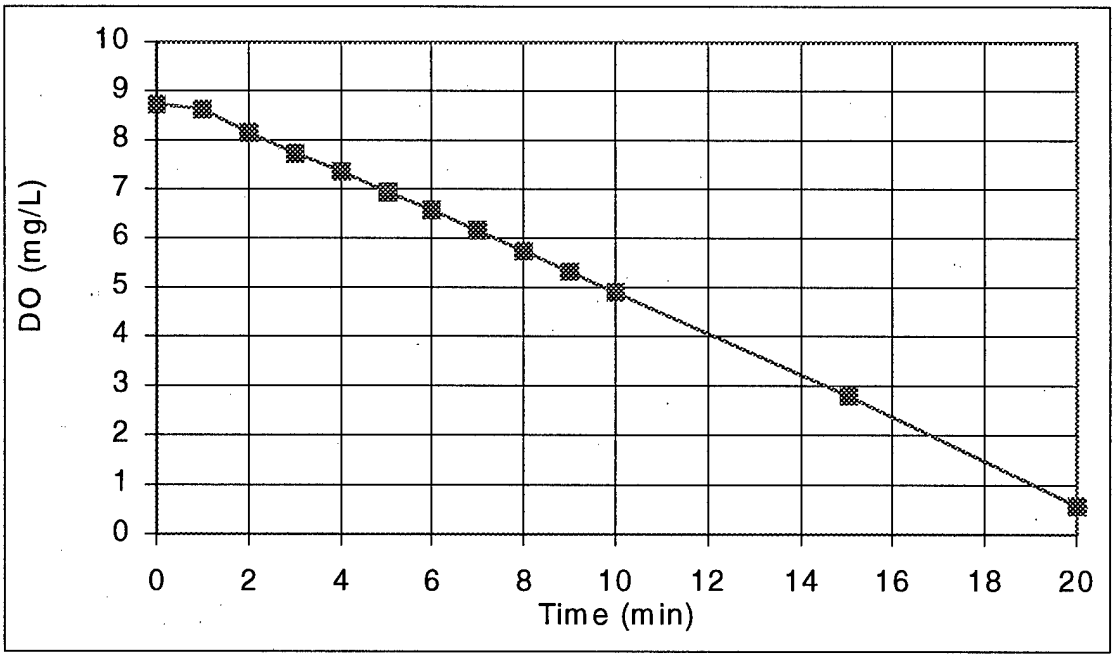

Figure 4.32 LOTT's OUR_{endog} Test Results
(MLVSS = 4190 mg/L, Temp = 17.7 °C)

Using the Microsoft Excel SLOPE function, the slope of the data over the linear oxygen utilization range was determined as -0.414 mg O_2/L-min with an endogenous OUR of 24.8 mg O_2/L-hr. The Specific Endogenous Oxygen Utilization Rate (SEOUR) was determined by dividing the OUR by the MLVSS concentration of 4190 mg/L, resulting in a SEOUR of 0.1396 g O_2/gVSS-d. The temperature during the test was 17.7 °C. The SEOUR rate was corrected to 20 °C using Equation 2.3 to give an $SOUR_{20}$ of 0.1491-g O_2/gVSS-d. Using the 1.42 g O_2 / g MLVSS equivalency ratio, the MLVSS decay coefficient was estimated as 0.21 g/g-d. The decay coefficient, like Olympus Terrace is much larger than the values reported in the literature.

Next, the endogenous nitrate utilization rate (NUR_{endog}) was determined for the same mixed liquor as outlined in Chapter 2. The results of the NUR_{endog} test are shown in Figure 4.33 below:

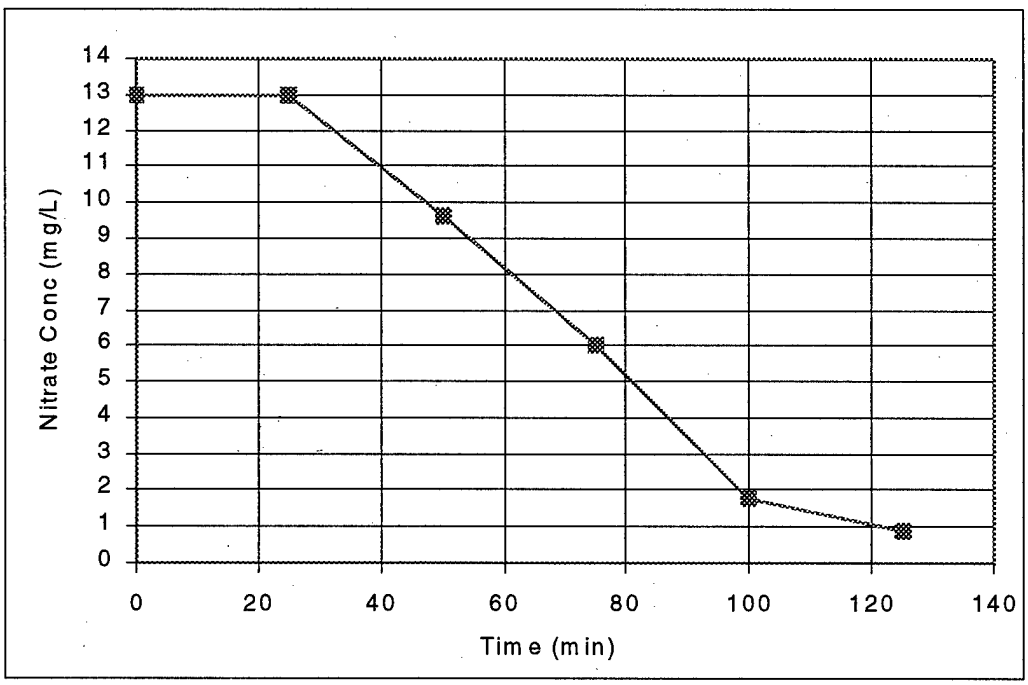

Figure 4.33 LOTT's NUR_{endog} Test Results
(MLVSS = 4190 mg/L, Temp = 24.7 °C)

A 25-minute lag time was noted before any appreciable nitrate removal took place. This lag time was shorter that the lag times observed in the Olympus Terrace and Snoqualmie Falls NUR_{endog} tests. However, this sample was allowed extra time to purge excess DO after the OUR test. This observation supported the acclimation findings noted earlier.

Using the Microsoft Excel SLOPE function, the slope of the data over the linear oxygen utilization range was as -0.1493 mg NO_3-N/L-min with an endogenous NUR of 8.96 mg NO_3-N/L-hr. The resulting $SDNR_{endog}$ was 0.0508 g NO_3-N/gVSS-d. The temperature during the test was 24.7 °C. The $SDNR_{endog}$ was corrected to 20 °C using Equation 2.3, to give an $SDNR_{endog20}$ of 0.0444 g NO_3-N/gVSS-d. This results in an equivalent specific oxygen uptake rate of (0.0444 g NO_3-N/gVSS-d)*(2.86 g O_2/g NO3-N) = 0.127 g O_2/gVSS-d.

The fraction of active biomass capable of nitrate reduction is thus estimated to be (0.127 g O_2/gVSS-d)/(0.1491 g O_2/gVSS-d) or 0.85. This denitrifying fraction is the largest of fractions measured for the four WWTPS. LOTT achieves an effluent TIN of less 3.0 mg/L and an effluent ortho-phosphate concentration averaging close to 3 mg/L. The high level of nitrate removal at this plant indicates favorable conditions for the growth of large denitrifying bacteria population fraction. The large anoxic tank volumes in comparison to the aerobic volumes coupled with the recycle ratio of 4:1 to the lst staged anoxic zone creates extensive anoxic conditions in comparison to the other WWTPs.

The SDNR based on the estimated active denitrifying biomass is shown in Figures 4.34 and 4.35 versus the reactor TBSCOD and RBCOD concentrations, respectively.

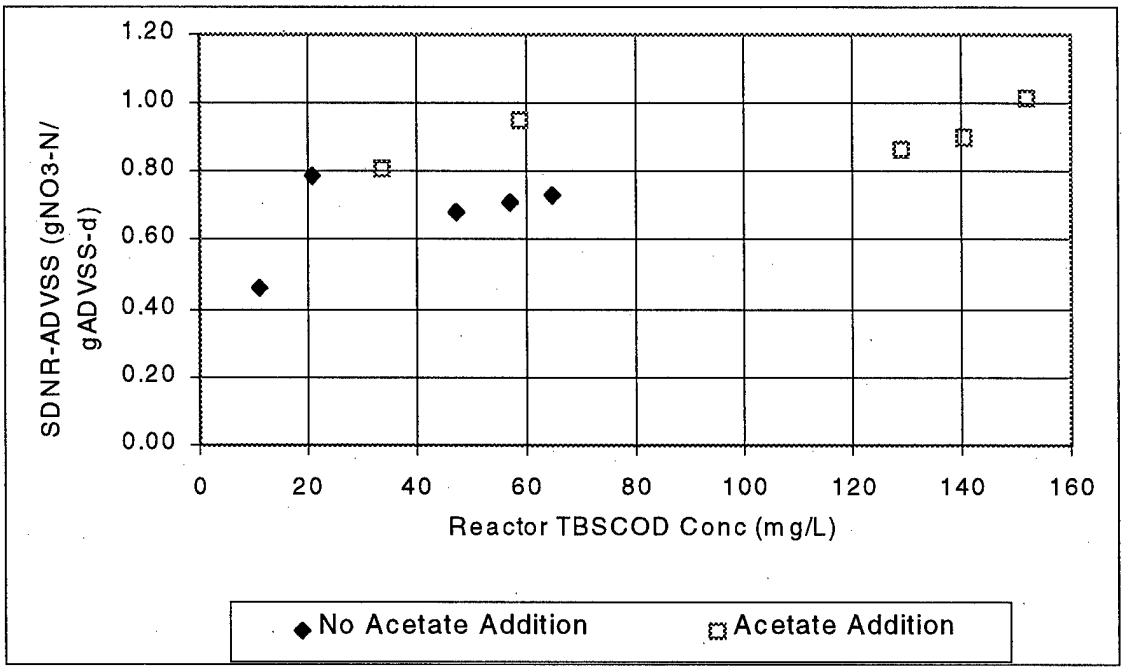

Figure 4.34 $SDNR_{ADVSS}$ (g NO3-N/gADVSS-d) versus Reactor TBSCOD Concentration (mg/L) for all LOTT Test Runs

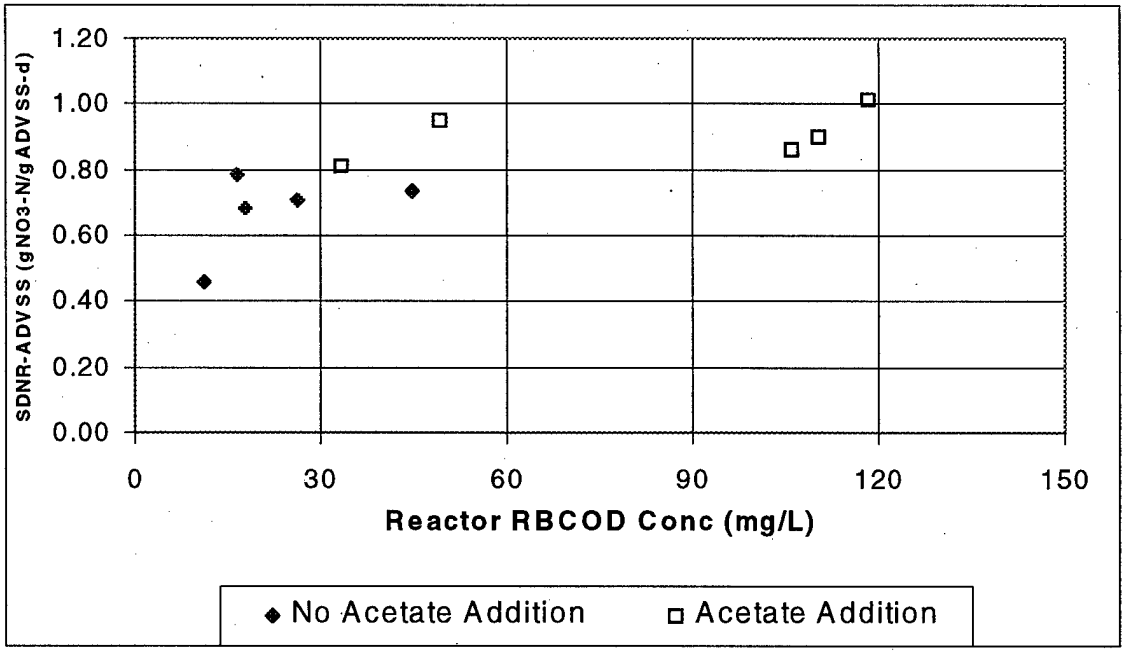

Figure 4.35 $SDNR_{ADVSS}$ (g NO3-N/gADVSS-d) versus Reactor RBCOD (mg/L) Concentration for all LOTT Test Runs

The corrected SDNRs are again directly proportional to the reactor TBSCOD and RBCOD concentrations. The SDNRADVSS varied from 0.461 to 1.013 g NO_3/g ADVSS-d. A maximum SDNR was 1.013 was observed.

4.3.4 COD and Nitrate Utilization

The specific substrate utilization rates during anoxic site tests conducted on September 22, 1998 are reported in Table 4.17.

Table 4.17 LOTT Specific COD Utilization Rates for Test Runs 7-10 (Based on Active Biomass)

Run	HRT	Reactor Soluble COD Concentrations (mg/L)			Specific COD Utilization Rates (gCOD/gAVSS-d)			Acetate
	(min)	TBSCOD	BSCOD	RBCOD	R_{TBSCOD}	R_{BSCOD}	R_{RBCOD}	Addition
7	59.3	11	0	11	2.02	0.76	2.26	N
8	14.7	21	4.3	16.7	4.79	2.13	2.66	N
9	63.9	33.5	0	33.5	2.03	1.17	0.86	Y
10	14	58.9	9.6	49.3	5.06	0.76	4.3	Y

Unreliable quantification of the influent COD fraction concentrations during runs 1 to 6 at LOTT resulted in their exclusion from the COD utilization analysis.

As shown in Table 4.17, simultaneous RBCOD and BSCOD utilization was also observed in the experiments conducted at LOTT. The observed specific COD utilization rates do not correlate with the reactor soluble COD concentrations for runs with and without acetate addition. Similar specific COD utilization rates are also observed for runs with similar of HRT, regardless of acetate addition. The SDNR results in Figure 4.35 showed that higher SDNRs occurred with higher reactor RBCOD concentration and that acetate addition increased the SDNR observed. This suggests that the COD/NO3-N consumption ratio was variable during the reactor tests.

Table 4.18 shows the estimated SDNR for endogenous respiration and SDNR for RBCOD removal.

Table 4.18 Calculated COD/Nitrate Removal Consumption Ratios and applied F/M Ratios based on Reactor TBSCOD Concentration during LOTT Test Runs

RUN	HRT	MLSS	Endog Nitrate Usage	Observed SNDR	SDNR for COD removed	Spec TBSCOD	COD/ NO3	F/M Ratio
				ADVSS		Util		
	(min)	(mg/L)	(g/g-d)	(mg/L)	(g/gAVSS-d)	(g/g-d)	(g/g)	(g/g-d)
7	59.3	1227	0.0165	0.461	0.445	2.02	4.5	1.45
8	14.7	1280	0.0165	0.786	0.770	4.79	6.2	2.33
9	63.9	1453	0.0165	0.811	0.795	2.03	2.6	3.84
10	14	1580	0.0165	0.95	0.934	5.06	5.4	8.7

An estimated COD/NO3-N ratio was calculated from the estimated active biomass yield of 0.31 g VSS/g COD as discussed in section 4.1.5 resulting in a ratio of 5.1 g COD/g NO3-N used. The COD/NO3-N consumption ratios for runs 7,8,10 are all close to the theoretical value. The COD/NO3 ratio of 2.6 for Run 9 was obtained for the reactor operating at longer HRT. This again suggests that particulate COD may have been used for denitrification. The F/M ratios shown also indicate relatively high substrate loading conditions were observed within the anoxic zone.

4.5 Review of the Test Results from all Experiments Conducted

4.5.1 SNDR Observations and Results

A summary of the observed SDNRs versus test HRTs for all the site tests runs are shown in Figure 4.36.

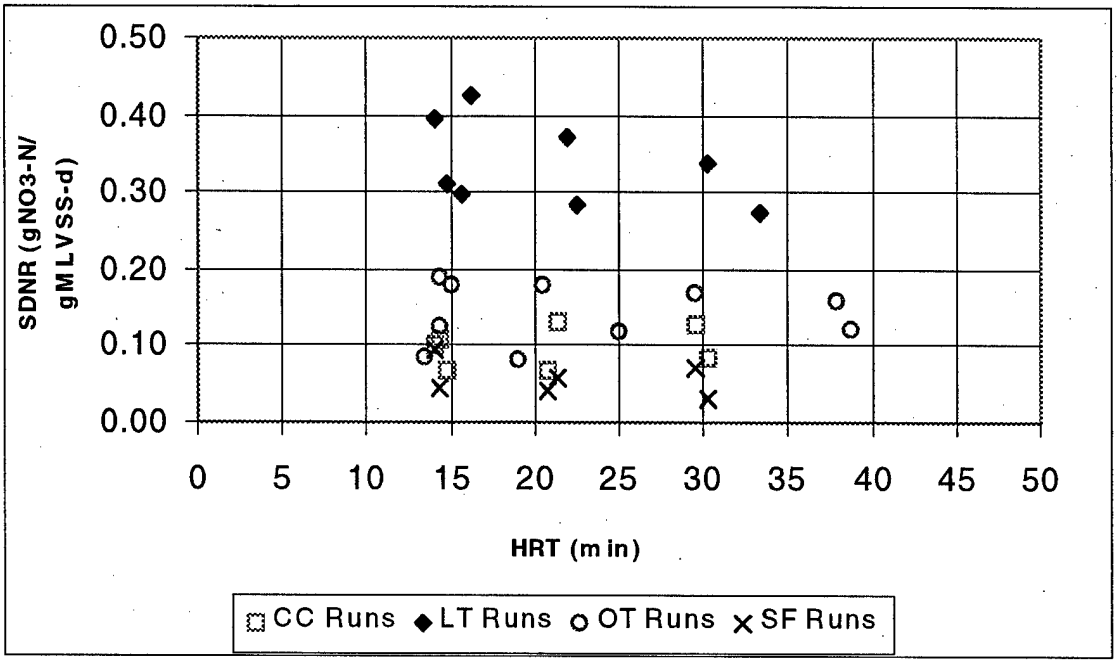

Figure 4.36 Observed SNDR (g NO3-N/g VSS-d) versus Reactor HRT (mg/L) for all Experimental Runs

These SDNRs show no correlation with HRT but are shown here to indicate that the test observed SDNRs are within the range of reported values in the literature. The HRTs shown in these test results are the actual hydraulic retention times and would be comparable to the nominal retention times (NRTs) of full-scale facilities where the NRT is the reactor value divided by the influent flow plus recycle flow rates. The recycle flow rate to an anoxic reactor in an MLE system can be 2-3 times the influent flow rate. To put the influent numbers in Figure 4.36 in perspective, the HRT should be multiplied by a factor of 3 or 4 to represent HRTs of actual MLE anoxic reactors. Nevertheless these results clearly show that the SDNRs for anoxic reactor must be related to fundamental bio-kinetic parameters that effect reaction rates. In this research the SDNRs were compared to the applied F/M ratio and soluble RBCOD concentrations.

A comparison between the temperature corrected SDNRs to 20 °C and applied F/M ratio, based on total biodegradable soluble COD loading and unadjusted for active biomass is shown in Figure 4.37.

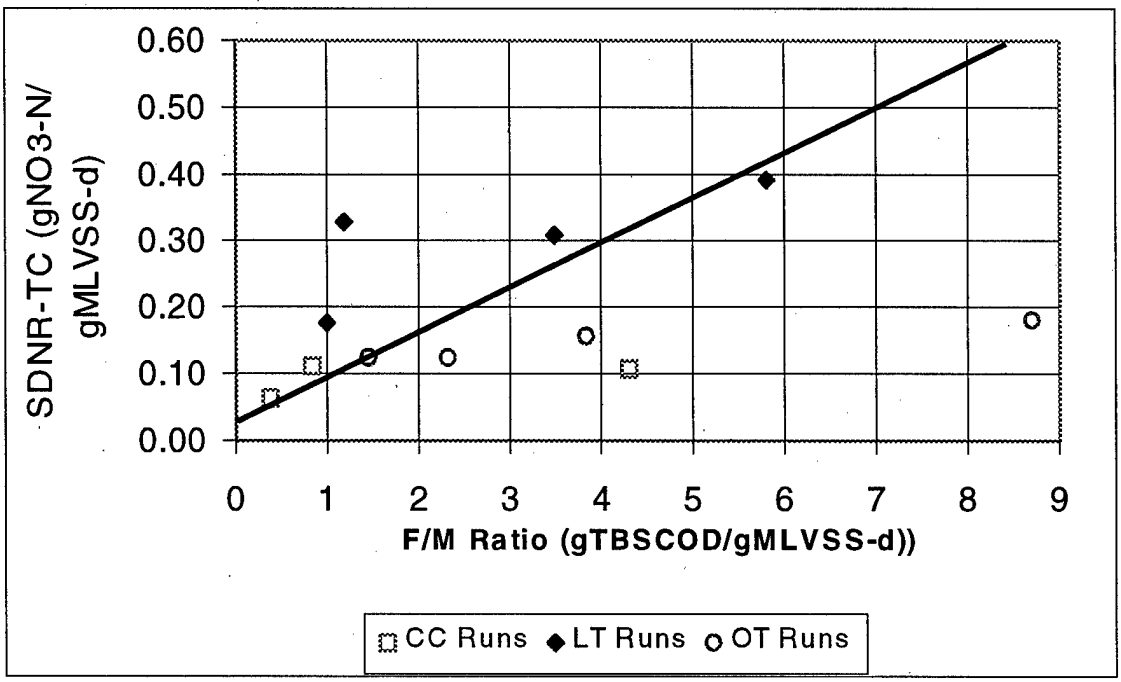

Figure 4.37 Observed $SNDR_{TC}$ (g NO_3-N/g VSS-d) versus F/M Ratio (gTBSCOD/gVSS-d) for all Experimental Runs

The solid line represents the SDNR to F/M ratio relationship predicted by Equation 4.4. It should be noted that the predicted SDNRs from the F/M equation are only approximate since the same relationship between total BOD_5, total degradable COD and percent soluble COD was assumed for all four plants. The data shows that only at the lower F/M ratios did the predicted SDNR more consistently match the measured SDNRs. The higher SDNRs near a F/M ratio of 1.0 are for the LOTT plant that had a higher influent soluble COD concentration. The lower F/M values are in the range of design loading used for single staged anoxic reactor with detention times of 3 to 4 hours. Thus it appears that the F/M equation approach provided conservative estimation of SDNRs for longer HRT anoxic reactors. However, the equation is inadequate for the design of staged reactors that would have higher F/M ratios in the initial stages and it may underestimate SDNRs for wastewaters with a higher fraction of RBCOD.

In the following sections the SDNRs are summarized and discussed as a function of the reactor RBCOD concentration and the denitrifying active biomass.

4.5.2 Denitrifying Biomass Fraction

Table 4.19 summarizes the estimated active denitrifying fraction for the mixed liquors of each of the four plants along with their reported operational SRTs.

Table 4.19 Estimated Denitrifying Fractions and Reported Operational SRTs for each of the four WWTP Mixed Liquors Used

	Sludge Age	*Denitrifying*
Plant	***(days)***	***Fraction***
Olympus Terrace	13	0.52
Snoqualmie Falls	23	0.67
Chambers Creek	3.6	0.30
LOTT	11.6	0.85

A wide range of denitrifying biomass fractions was observed ranging from 0.30 for Chamber's Creek to 0.085 for LOTT. All the WWTPs had a larger denitrifying fraction than the BioWin model's default fraction of 0.37, except Chamber's Creek. As mentioned earlier, LOTT does achieve a high level of nitrogen and phosphorus removal. Chamber's Creek reports minimal nitrogen removal in comparison with effluent TINs above 20 mg/L. The fraction is not correlated with system SRTs and the reasons for the variation are not obvious.

4.5.3 Summary of Active Biomass

Composite results from the active biomass quantification for all plants are shown in Table 4.20:

Table 4.20 Summary of Active Biomass Fractions determined for the Mixed Liquor at each of the four WWTPs

	Primary Treatment	*VSS/TSS*	*SRT*	*Active*
Plant		*Ratio*	*(days)*	*Fraction*
Olympus Terrace	N	0.76	13	0.39
Snoqualmie Falls	N	0.81	23	0.30
Chambers Creek	Y	0.76	3.6	0.44
LOTT	Y	0.79	11.6	0.46

The estimated active biomass comprised less than 50% of the mixed liquor suspended solids for all four WWTPs and varied from 0.30 to 0.46 even though the mixed liquor VSS/TSS varied within a smaller range of 0.76 to 0.81. The variation in active mass fraction followed expected trends. The plants without primary treatment had lower active mass fractions. Without primary treatment more non-biodegradable solids would be contained in the aeration tank mixed liquor to thus lower the active mass fraction. The ability of having an accurate estimate of the active biomass as well as knowing the fraction of active biomass capable of denitrification is important in understanding and applying denitrification kinetics for anoxic reactor design and to evaluate performance. The inert VSS is expected to vary in the influent flow for different WWTPs and thus must be determined to develop more accurate design and more accurate assessments of plant performance.

4.5.4 Evaluation of Denitrification Kinetics

The anoxic test reactor results from all four WWTPs showed that the SDNRs based on active denitrifying biomass are related to the RBCOD and the relationship follows a Michaelis-Menten type curve. The IAWQ model also assumes that the SDNRs should be correlated with RBCOD concentrations in that way. The model structure assumes that degradable particulate COD and slowly biodegradable COD is converted to "true soluble" readily biodegradable (RBCOD) and that the RBCOD utilization follows Michaelis-Menten kinetics also. The model also shows that the nitrate utilization rate is directly

proportional to the RBCOD utilization rate. The observed SDNR can be described as follows based on a nitrate demand for endogenous respiration and for oxidation of RBCOD:

$$SNDR = SDNR_{endog} + \frac{A}{2.86} * \left(\frac{K * RBCOD}{Ks + RBCOD} \right) \qquad (4.14)$$

Where:

SDNR = Observed specific denitrification rate, g NO_3-N/gADVSS-day

A = g O_2 used per g RBCOD utilized

K_s = half-velocity coefficient, mg/L

K = maximum specific RBCOD utilization rate, g RBCOD/gADVSS-day

The endogenous SDNR can be estimated from the endogenous decay rate of the active denitrifying biomass.

$$SDNR_{endog} = \frac{1.42 * k_d}{2.86} \qquad (4.15)$$

Where:

k_d = endogenous decay coefficient, g COD endogenous mass/g COD active mass-day

A commonly used estimated value for k_d is of 0.2 g/g-d, which would yield an $SDNR_{endog}$ of 0.10 g NO_3-N/d ADVSS-d. $SDNR_{ADVSS}$ versus reactor RBCOD curve shown for all the WWTP site tests is shown in Figure 4.38.

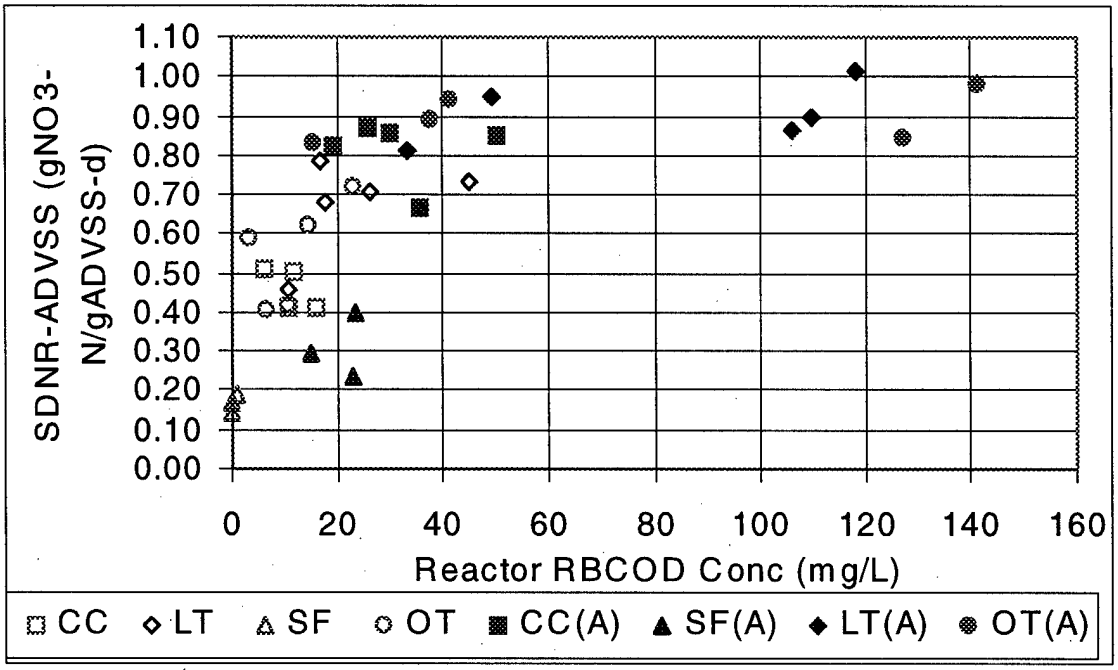

Figure 4.38 $SDNR_{ADVSS}$ (gNO_3-N/gADVSS-d) versus reactor RBCOD Concentration (mg/L) for all four WWTP Runs

Solid symbols in Figure 4.38 represent runs with acetate addition. The lowest SNDR values were from Snoqualmie Falls runs without acetate addition where reactor RBCOD and BSCOD were not detected. The data in Figure 4.37 does not suggest that the kinetics with acetate addition are different than with the wastewater, so all of the data is evaluated together for the denitrification kinetic evaluation. Curve fitting software was used to find the 'best-fit' curve for the data in Figure 4.37. The software used was called Curve Expert 1.3. This software has a function called curve finder that automatically checks data inputted on a clipboard for the best curve fit. This function shifts through every possible curve using both linear and nonlinear regressions and ranks the fit of each method from best to worst fit. When the curve finder function was used, the top five curve fits were exponential association, a Sigmoidal family or 'S' shaped growth model, rational function, polynomial (4^{th} degree) fit, and logistic fit. The results for the exponential association fit are shown in Figure 4.39.

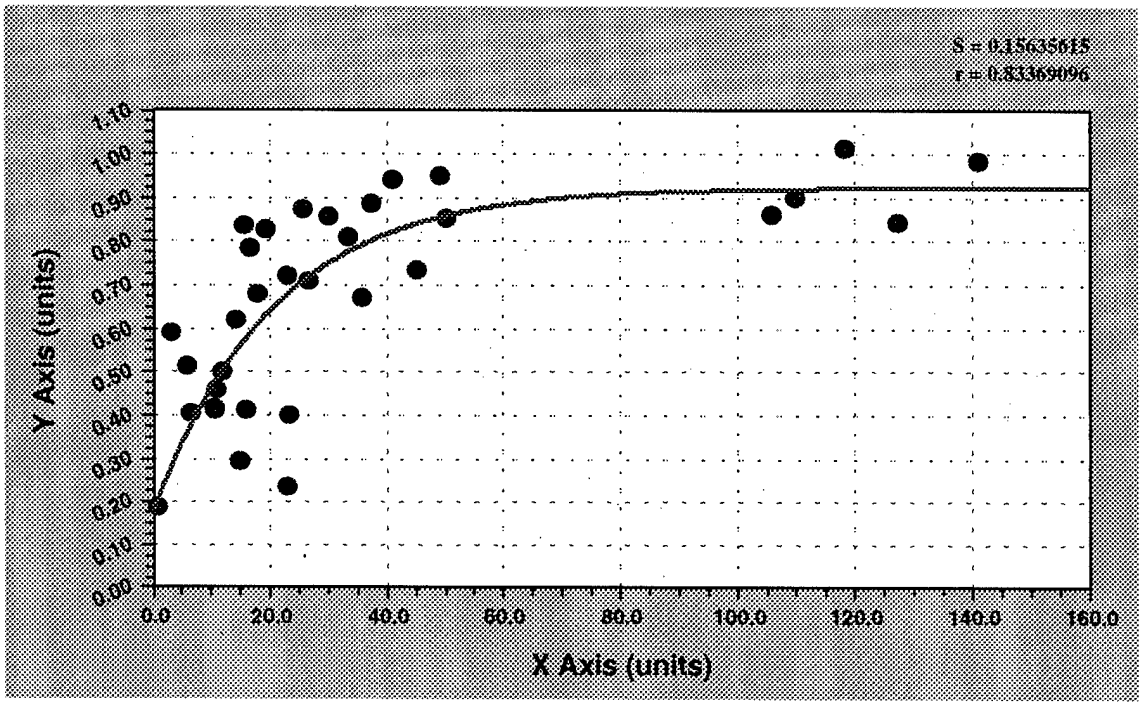

Figure 4.39 Curve Fit for Data in Figure 4.37 where X-Axis is Reactor RBCOD Concentration (mg/L) and Y-axis is the $SDNR_{ADVSS}$ (g NO3-N/gADVSS-d)

The fit converged to a tolerance of 1e-006 in 4 iterations with a standard error (s) of 0.156 and a correlation coefficient (r) of 0.8337. The exponential association used the following equation to fit the curve: $y=a(b-e^{-cx})$. A Y-intercept of 0.19 is found with this curve fit indicating a minimum or endogenous SDNR of 0.19 g NO_3-N/g ADVSS-d. This value is just more than four times as large the theoretical $SDNR_{endog}$ of 0.04 g NO_3-N/g ADSS-d determined using Equation 4.15 but is still reasonable and was used from this point on as the $SDNR_{endog}$ rate.

The data was corrected for the estimated $SDNR_{endog}$ by subtracting 0.19 from each data point. All of the $SDNR_{ADVSS}$ values for the WWTP tests minus the $SDNR_{endog}$ rate of 0.19 are plotted in Figure 4.40 as a function reactor RBCOD concentration.

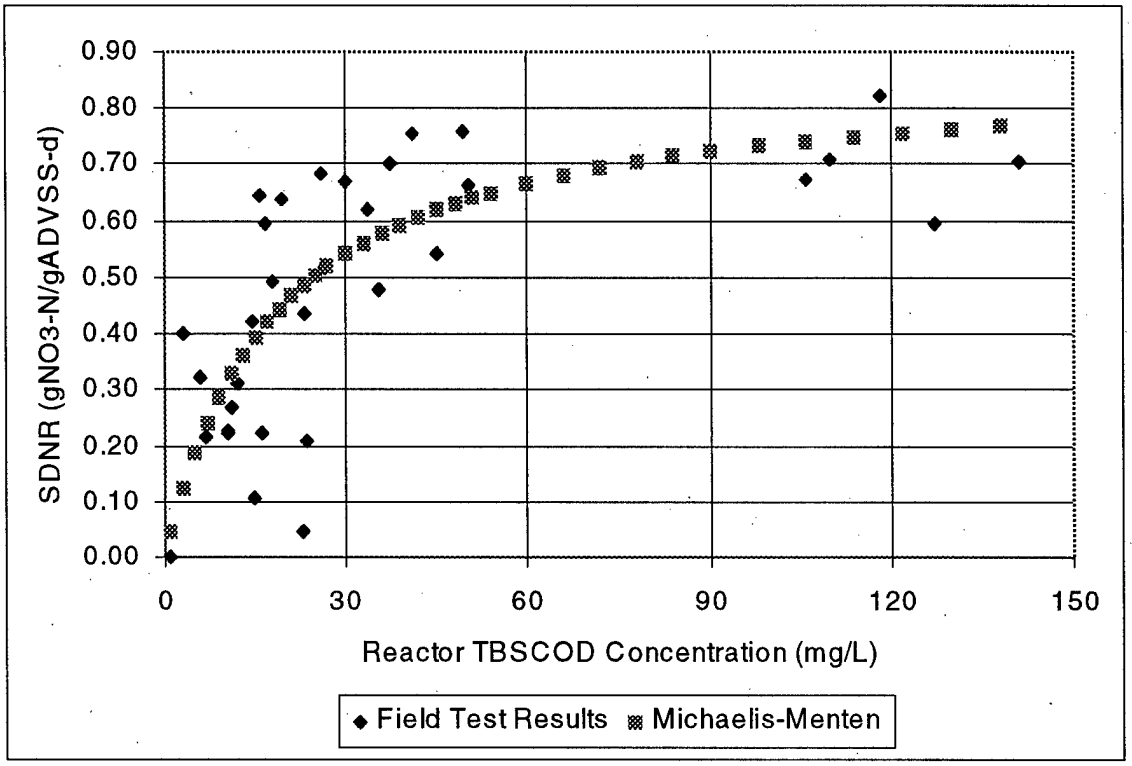

Figure 4.40 $SDNR_{ADVSS}$ (g NO_3-N/g ADVSS-d) versus Reactor RBCOD Concentration (mg/L) for all WWTP Test Runs

A statistical fit of the corrected data was made using the Curvefit program and a standard Michaelis-Menten growth model. Curvefit yielded a maximum SDNR of 0.868 g NO3-N/gADVSS-d and half velocity constant of 18.1 mg/L. The correlation coefficient of the fit was 0.827 with a standard error of 0.151. This K_s value of 18.1 mg/L is significantly higher than the default value of 5.0 mg/L used in the IAWQ model. However, the data shows variability of SNDRs for given RBCODs as effected by plant wastewater characteristics.

The database from the plants studied here provide a useful guideline for a k_s value for RBCOD substrate utilization with denitrification. Calibration of kinetics for a specific plant may yield a different k_s value. A maximum specific substrate utilization rate may also be estimated from the data fit. The maximum SDNR can be estimated as equal to

AK/2.86 in Equation 4.14. To determine K, the value for A was calculated from the default yield value of 0.666 (g biomass COD/g RBCOD used) in the IAWQ model for biomass growth under anoxic conditions by the following Equation.

$$A = 1.0 - Y \qquad (4.16)$$

Where:

Y = g biomass as COD produced/g RBCOD used

Equation 4.16 yielded a value for A of 0.334 g/g and a maximum specific substrate utilization (K) value of 7.43 gCOD/gAVSS-d. The maximum specific growth rate is calculated as follows:

$$\mu_m = YK = (0.666)(7.08) = 4.95 \text{ g/g-d} \qquad (4.17)$$

This compares to a default value of 3.2 g/g-d in the IAWQ model. Based on variability of the experimental data and all the assumptions made, the correlation with IAWQ default is extremely good. A comparison of the experimental data kinetics and the IAWQ model bacteria growth kinetic coefficients is made in Figure 4.41.

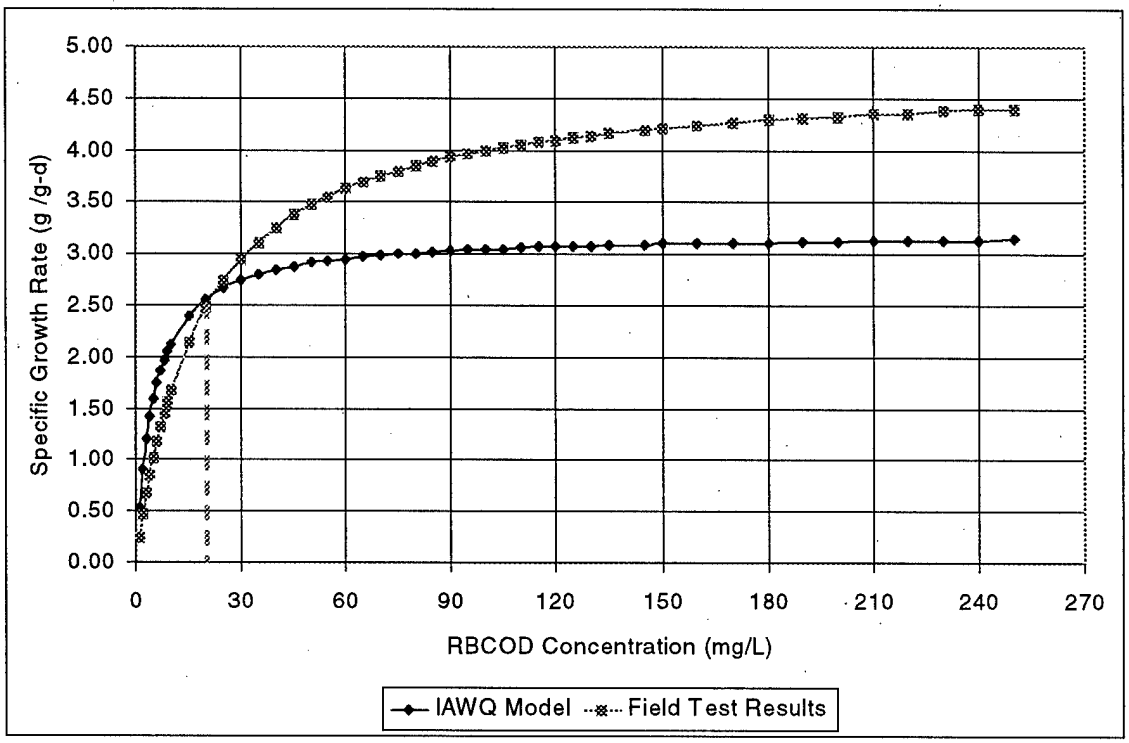

Figure 4.41 Graph of Monod Model for RBCOD Degradation Using IAWQ Model Coefficients and Coefficients from Field Denitrification Tests

At RBCOD concentrations of <20 mg/L the IAWQ model default and the estimated growth coefficients predict very similar specific growth rates. Even though the maximum specific growth rate coefficient is higher for the experimental data than the IAWQ model's coefficient, the lower k_s for the IAWQ models predicts slightly higher growth rates at low RBCOD concentrations. The specific substrate utilization rate would be directly proportional to the specific growth rates.

These kinetic coefficients for denitrification kinetics are based on a significant database. No significant database has been shown to support the IAWQ model for RBCOD substrate utilization under anoxic conditions. However a close agreement between these results and the default IAWQ growth rates was shown.

A practical issue in applying these kinetics however is the need to predict the fraction of active heterotrophic biomass capable of denitrification (η_g). The IAWQ default value of 0.37 may provide a conservative design based on the results of this study since three of the four plants in this study had η_g values greater than 0.37.

4.5.5 Substrate Utilization

Composite COD to nitrate utilization rates from all September experiments is shown in Table 4.21:

Table 4.21 Calculated COD/Nitrate Removal Consumption Ratios and F/M Ratios for all Experimental Runs

RUN #	HRT	MLVSS	Endog Nitrate	Observed SNDR	SDNR for	Spec TBSCOD	COD/ NO3	F/M Ratio
			Usage	ADVSS	COD rem	Util		
	(min)	(mg/L)	(g/g-d)	(g/g-d)	(g/g-d)	(g/g-d)	(g/g)	(g/g-d)
OT7	38.8	1550	0.0165	0.609	0.593	9.27	15.6	1.45
OT8	14.4	1827	0.0165	0.625	0.609	7.52	12.3	2.33
OT9	37.8	1985	0.0165	0.783	0.767	6.95	9.1	3.84
OT10	15.1	1984	0.0165	0.851	0.835	6.48	7.7	8.7
CC7	62.5	1673	0.0165	0.503	0.487	1.31	2.69	0.37
CC8	62.5	1800	0.0165	0.874	0.858	3.97	4.62	0.84
CC9	14.4	1826	0.0165	0.851	0.835	6.79	8.1	4.3
LT7	59.3	1227	0.0165	0.461	0.445	2.02	4.5	1.45
LT8	14.7	1280	0.0165	0.786	0.770	4.79	6.2	2.33
LT9	63.9	1453	0.0165	0.811	0.795	2.03	2.6	3.84
LT10	14	1580	0.0165	0.95	0.934	5.06	5.4	8.7

No clear general relationship was established between F/M ratio and specific substrate utilization and specific nitrate removal rates. If 20% of the COD was used in synthesis, the ratio would be 3.6 gCOD/gNO3-N, which is close to the value of 3.45 reported by Randall et al. (1992) from experimental tests. LOTT's COD/NO_3-N data and some of Chamber's Creek data fall within the reasonable range of COD/NO_3-N ratios. As discussed earlier the effects of particulate COD utilization and/or cell COD storage cannot be accounted for in these experiments. No meaningful conclusions can be drawn on the COD/NO3-N ratio developed in these experiments.

Figure 4.42 shows a plot of COD/NO_3-N consumption ratio versus the F/M ratio during the anoxic reactor tests.

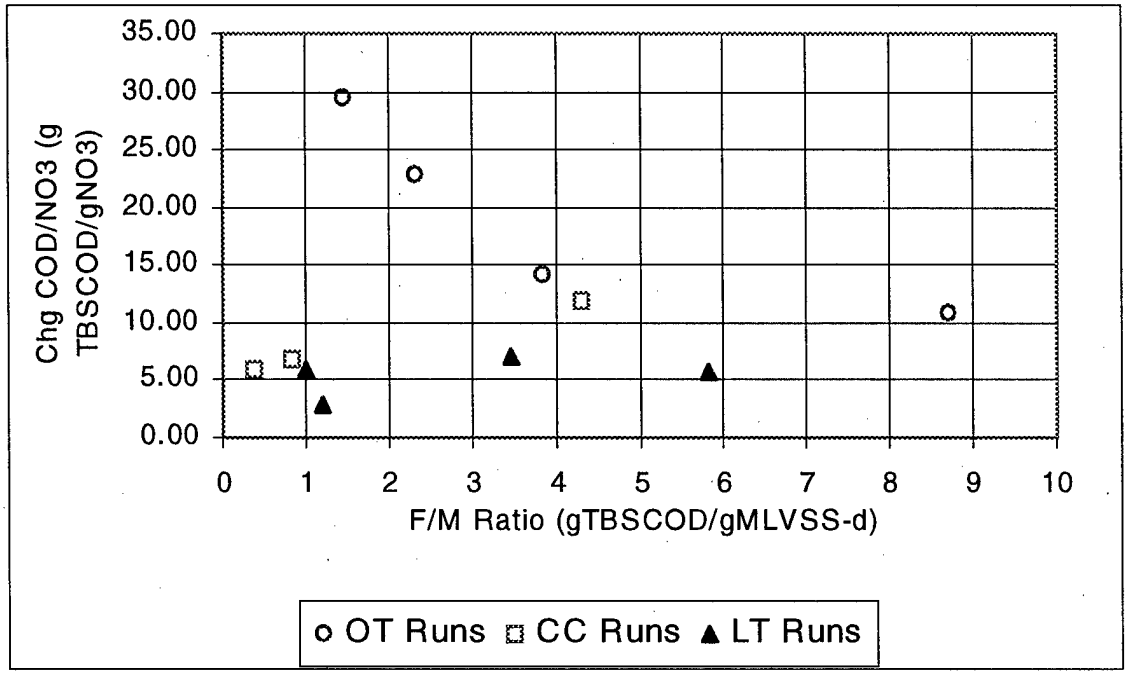

Figure 4.42 Observed COD/NO3-N Use (g/g-d) versus F/M (gTBSCOD/gMLVSS-d) for all Experimental Runs

Excluding the results of above 10.0 g/g, an average of 6.3 g O_2/gSCOD removal or about 45% of the COD is oxidized and about 55% is incorporated into the cell mass. This yield of 0.55 g/g-d is close to the yield value 0.666 g/g-d given in the IAWQ model. This analysis does not consider the COD obtained from the degradable particulate COD. This may be small but would result in a higher cell yield value.

Chapter 5 Summary and Conclusions

The experiments conducted during this research examined factors that affect specific denitrification rates in municipal wastewater treatment plants biological nitrogen removal processes. The effects of influent wastewater characteristics and the amount of dissolved organic substrate (termed readily biodegradable COD or RBCOD), and the effects of the anoxic zone operating conditions on SDNRs were examined. Studies were carried out at four municipal WWTP sites in the Seattle area to obtain a range of SDNRs for different wastewater and activated sludge conditions. SDNRs were observed within a 2-liter anoxic reactor operated at the sites in which the wastewater and activated sludge mixed liquor were contacted. The feed flow rate to the reactor was varied to observe the affects of different substrate loading conditions. Acetate was added to the influent wastewater in some tests to observe the effects of higher RBCOD loading conditions within the reactor. Plant data on operating SRT, influent BOD, and total solids production were evaluated to estimate the active biomass fraction of each plant's mixed liquor. In addition, laboratory oxygen and nitrate endogenous uptake rate tests were conducted to determine the fraction of the active mass capable of denitrifying. The following conclusions were developed from this research.

1. Once the mixed liquor for the four WWTPs was defined in terms of active denitrifying biomass, the combined SDNR (corrected to 20 °C for the plants) could be modeled as a function of the RBCOD using a Michaelis-Menten kinetic model. A maximum specific RBCOD uptake rate of 7.43 g RBCOD/gADVSS-d was estimated. Interpretation of the Michaelis-Menten type curve yielded a half velocity coefficient of 18.1 mg/L and a maximum specific growth rate of 4.95 g/g-d.

2. The active biomass fraction of the mixed liquor varied between the four WWTPs, ranging from 0.30 to 0.46. The two plants without primary clarification had lower active

biomass fractions ranging from 0.30 to 0.39. The plants with primary clarification had higher active biomass fractions ranging from 0.44 to 0.46.

3. The fraction of the mixed liquor active biomass used from the four WWTPs test sites capable of denitrification varied from 0.30 to 0.85. The lower fraction corresponded to a plant achieving low levels of total nitrogen removal while the higher fraction corresponded the WWTP achieving a high level of nitrogen removal with an effluent TIN below 3.0 mg/L.

4. When the mixed liquor from the WWTPs was stored in a cold room overnight and then aerated for four to five hours at room temperature, a lag time of 25 to 40 minutes was observed before nitrate reduction occurred.

5. In all anoxic reactor experimental runs concurrent utilization of the readily biodegradable soluble COD (RBCOD) and the slowly biodegradable soluble COD (SBCOD) was observed. This supports the IAWQ assumption that SBCOD use where the SBCOD is hydrolyzed to RBCOD and then the RBCOD is oxidized by the active biomass.

6. The commonly used equation that relates SNDRs to the anoxic zone F/M ratio is of limited accuracy. The equation provided conservative estimates of SDNRs at low F/M loading condition (<1.0). However, the equation is inadequate for the design of staged reactors that would have higher F/M ratios in the initial stages and it may also underestimate SDNRs for wastewaters with a higher fraction of RBCOD.

7. The large difference in denitrifying fractions shown in this study indicate that the default parameter value of the active heterotrophic active bimass fraction capable of denitrification, η_g, in the IAWQ model can vary widely for different wastewater treatment plants.

REFERENCES

Albertson, O.E. and Jim Coughenour (1995). "Aerated Anoxic Oxidation-Denitrification Process", *Journal of Environmental Engineering*, 121(10), 720-726.

Albertson, O.E., and P. Hendricks (1992). "Bulking and Foaming Organisms Control at Phoenix AZ WWTP", *Water Science Technology*, 26(3-4), 461-472.

Bang, D.Y., Y. Watanabe and T. Noike (1995). "An Experimental Study on Aerobic Denitrification with Polyvinyl Alcohol as a Carbon Source in Biofilms". *Water Science Technology*, Vol 22(8), 235-242.

Barber, John B. and Michael Bullard (1994). "Compensating for Limited Aeration Capacity with Anoxic Pretreatment", *Water Environment Federation 67th Annual Conference & Exposition*, 175-185.

Barker, P.S. and P.L. Dold (1995). "COD and Nitrogen Mass Balances in Activated Sludge Systems", *Water Research*, 29(2), 633-643.

Barker, P.S. and P.L. Dold (1997a). "General Model for Biological Nutrient Removal Activated Sludge Systems: Model Application", *Water Environment Research*, 69(3), 001-007.

Barker, P.S. and P.L. Dold (1997b). "General Model for Biological Nutrient Removal Activated Sludge Systems: Model Presentation", *Water Environment Research*, 69(3), 008-016.

Bradstreet, Kenneth A. and Gary R. Johnson (1994). "Study of Critical Operational Parameters for Biological Nitrogen Reduction at a Municipal Wastewater Treatment Plant", *Water Environment Federation 67th Annual Conference & Exposition*, 669-680.

Bortone, G., R. Saltarelli, V. Alonso, R. Sorm, J. Wanner, A. Tilche (1996). "Biological Anoxic Phosphorus Removal – The Dephanox Process", 18th IAWQ Biennial International Conference. Singapore, 102-109.

Burdick, Chuck R., David R. Refling, and H. David Stensel (1982). "Advanced Biological Treatment to Achieve Nutrient Removal", *Journal of Water Pollution Control Federation*, 54(7).

Clayton, J.A., G.A. Ekama, M.C. Wentzel, and G.v.R. Marais (1991). "Denitrification Kinetics in Biological Nitrogen and Phosphorus Removal Activated Sludge Systems Treating Municipal Wastewaters', *Water Science Technology* 416(23), 1025-1035.

Crites, Ron and George Tchobanoglous (1998). Small and Decentralized Wastewater Management Systems. WCW McGraw-Hill Companies, Inc.

Ekama, G.A., P.L. Dold, and G.v.R. Marais (1986). "Procedures for Determining Influent COD Fractions and the Maximum Specific Growth Rate of Heterotrophs in Activated Sludge Systems". *Water Science Technology*, 18(6), 91-114.

Eliosov, Boris, David Rubin, Vyacheslav Libman, and Yerachmiel Argaman (1997). "A New Approach to the Effects of Dissolved Oxygen on the Rate of Denitrification", *Water Environment Federation 70th Annual Conference & Exposition*, 325-332.

Filipe, Carlos D. and Glen T. Daigger (1997). "Evaluation of the Capacity of Phosphorus Accumulating Organisms to use Nitrate as well as Oxygen as Final Electron Acceptor: A Theoretical Study on Population Dynamics", *Water Environment Federation 70th Annual Conference & Exposition*, 341-352.

Fries, M. Kim, Barry Rabinowitz, and Ribert N. Dawson (1994). "Biological Nutrient Removal at the Calgary Bonnybrook WWTP', *Water Environment Federation 67th Annual Conference & Exposition*, 643-656.

Garber, James B., Shin Joh Kang, Gary L. Hill, and Oneil Wilson (1994). "The Use of Anoxic Selector Activated Sludge for Biological Nutrient Removal and Filamentous Growth Control at a Tobacco Processing Facility", *Water Environment Federation 67th Annual Conference & Exposition*, 411-417.

Hao, Oliver J., and Michael H. Kim (1990). "Continuous Pre-Anoxic and Aerobic Digestion of Waste Activated Sludge", *Journal of Environmental Engineering*, 116(5), 863-879.

Henze, M (1991). "Capabilities of Biological Nitrogen removal Processes from Wastewater", *Water Science Technology*, 23(Kyoto), 669-679.

Hong, Sun-Nan, Feng-Ying Chang, and R. David Holbrook (1997). "Enhancing Denitrification in the Secondary Anoxic Zone by RAS Addition: A Full Scale Evaluation, *Water Environment Federation 70th Annual Conference & Exposition*, 411-417.

Jain, R., G. Lyberatos, S.A. Svoronos, and B. Koopman (1992). "Operational Strategies for Pre-denitrification Process", *Journal of Environmental Engineering*, 118(1), 56-67.

Kerrn-Jespersen, J.P.and M. Henze (1993). "Biological Phosphorus Uptake Under Anoxic and Aerobic Conditions", *Water Resources*, 27(4), 617-624.

Kjaergaard, L (1977). "The Redox Potential: Its Use and Control in Biotechnology", *Advances in Biochemical Engineering*, 131-150.

Lie, Ewa, and Thomas Welander (1994). "Influence of Dissolved Oxygen and Oxidation-Reduction Potential on the Denitrification Rate of Activated Sludge", *Water Science and Technology*, 30(6), 91-100.

Ishizaki, A., H. Shibia, and Y. Hirose (1974). "Basic Aspects of Electrode Potential Change in Submerged Fermentation", *Journal of Agriculture and Biological Chemistry*, 38(12), 2399-2406.

Lo, C.K., C.W. Yu, N.F.Y. Tam, and S. Traynor (1994). "Enhanced Nutrient Removal by Oxidation-Controlled Aeration in a Laboratory Scale Extended Aeration Treatment System", *Water Research*, 28(10), 2087-2094.

Mamais, Daniel, David Jenkins, and Paul Pitt (1993). "A Rapid Physical-Chemical Method for the Determination of Readily Biodegradable Soluble COD in Municipal Wastewater", *Water Research*, 27(1), 195-197.

McClintock, Samuel A., Joseph H. Sherrard, John T. Novak, Clifford W. Randall (1988). "Nitrate Versus Oxygen Respiration in the Activated Sludge Process", *Journal of Water Pollution Control Federation*, 60(3), 342-350.

Melcher, Henryk, Patrica Tam, Larry Ekstrom, Rick Stilwill (1994). "Full Scale Experience with Biological Process Models – Calibration Issues", *Water Environment Federation 67th Annual Conference & Exposition*, 77-87.

Metcalf and Eddy (1991), Wastewater Engineering. Treatment, Disposal, and Reuse. McGraw-Hill, New York.

Mines, Richard O. Jr (1997). "Nutrient Removal Using a Modified Pure Oxygen Activated Sludge Process", *Advances in Environmental Research*, 1(1), pp. 15-26.

Moriyama, Katsumi, Kazuaki Sato, Yoshinobu Harada, Kazou Washiyama, and Koich Okamoto (1990). "Renovation of an Extended Aeration Plant for Simultaneous Biological Removal of Nitrogen and Phosphorus Using Oxic-Anaerobic-Oxic Process", *Water Science Technology*, 22(7/8), 61-68.

Munch,Elisabeth V., Paul Lant, and Jurg Keller (1996). "Simultaneous Nitrification and Denitrification in Bench-Scale Sequencing Batch Reactors", *Water Resources*, 30(2), 277-284.

Murakami, Cey and Roger Babcock Jr (1998). "Effect of Anoxic Selector Detention Time and Mixing Rate on Denitrification Rate and Control of Sphaerotilus Natans Bulking", *Water Environment Federation 71st Annual Conference & Exposition*, 89-98.

Nue, Ken (1995). "Enhanced Biological Nutrient Reduction: Effective Field Sampling and Process Monitoring can make a Difference", *Water Environment Federation 68th Annual Conference & Exposition*, 347-355.

Painter, H.A (1970). "A Review of the Literature on Inorganic Nitrogen Metabolism in Microorganisms". *Water Research*, 4(5) 393-450.

Peddie, Craig C., Donald S. Mavinic, and Christopher J. Jenkins (1990). "Use of ORP for Monitoring and Control of Aerobic Sludge Digestion". *Journal of Environmental Engineering*. 116(3), 461-471.

Rabinowitz, B. and W.K. Oldman (1986). "Excess Biological Phosphorus Removal in the Activated Sludge Process Using Primary Sludge Fermentation", *Canadian Journal of Civil Engineering*, 13, 345-351.

Radjai, M., R.T. Hatch, and T.W. Cadman (1984). "Optimization of Amino Acid Production by Automatic Self-Tuning Digital Control of Redox Potential", Biotechnology and Bioengineering Symposium, John Wiley and Cons, Inc, New York, NY.

Randall, Clifford W., James L. Barnard, and H. David Stensel (1992). Design and Retrofit of Wastewater Treatment Plants for Biological Nutrient Removal. Technomic Publishing Company, Inc., Lancaster, PA.

Reardon, Roderick D, Tom Kolby, and Milton Odo (1996). "The LOTT Nitrogen Removal Facilities: A First Year Evaluation. *Water Environment Federation 69th Annual Conference & Exposition*, 695- 705.

Shibai, H. (1974). "Simultaneous Measurement of Dissolved Oxygen and Oxidation – Reproduction Potentials in the Aerobic Culture", *Journal of Agriculture and Biological Chemistry*, 38(12), 2407-2411.

Siebritz IP, GA Ekama, G.v.R. Marais (1983). " A Parametric Model for Biological Excess Phosphorus Removal". Water Science Technology, 15(3/4), 127-152.

Standard Methods for the Examination of Water and Wastewater, 17th Edition (1989). American Public Health Association, Washington, D.C.

Stensel, H. David, Thomas E. Coleman, W. Brent Denham, Darrel Fleischman (1995). "Innovative Process Used to Upgrade Oxidation Ditch for Nitrogen Removal and SVI Control". *Water Environment Federation 68th Annual Conference & Exposition*, 591-602.

Stevens, G.M., J.L. Barnard, M.K. Fries, and L. Forty (1997). "Enhancing Anoxic P Uptake in BNR Process". *Water Environment Federation 70th Annual Conference & Exposition*, 493-501.

Takeuchi, Jun-ichi (1991). "Influence of Nitrate on the Bacterial Flora of Activated Sludge under Anoxic Condition". *Water Science Technology*, 23 (Kyoto), 765-772.

Urbain, Vincent and Jacques Maniem (1997). "Potential of In Situ Volatile Fatty Acids Production as Carbon Source for Denitrification". *Water Environment Federation 70th Annual Conference & Exposition*, 333-339.

U.S. Environmental Protection Agency, (1993). Manual-Nitrogen Control, EPA/625/r-93/010.

Van Haandel, A.C., G.A. Ekama, and G.v.R. Mamais (1981). "The Activated Sludge Process-3". *Water Research*, 15, 1135-1152.

Wagner, Michael, Gabriele Rath, Hans-Peter Koops, Jannie Flood, and Rudolf Amann (1996). "*In Situ* Analysis of Nitrifying Bacteria in Sewage Treatment Plants", *Water Science Technology*, 34(1-2), 237-244.

Wareham, David G., Kenneth J. Hall, and Donald S. Mavinic (1993). "Real-Time Control of Aerobic-Anoxic Sludge Digestion Using ORP". *Journal of Environmental Engineering*. 119(1) 120-137.

Warner, A.P.C., G.A. Ekama, and G.v.R. Marias (1986). "The Activated Sludge Process – IV", *Water Research*, 20(8), 943-958.

Wentzel, M.C., G.A. Ekama, and G.v.R. Marais (1991). "Kinetics of Nitrification Denitrification Biological Excess Phosphorus Removal System – A Review", *Water Science Technology*, 23(Kyoto), 555-565.

Wentzel, M.C., and G.A. Ekama (1996). "Principals in the Design of Single Sludge Systems for Biological Removal of Carbon, Nitrogen, and Phosphorus". *Water Environment Federation 69th Annual Conference & Exposition*, 681-694.

Zaho, H.W., W.K. Mavinic, and F.A. Koch (1996). "Pre-Denitrification with Methanol in Single-Sludge Biological Nutrient Removal Processes Treating Domestic Sewage", *Water Environment Federation 69th Annual Conference & Exposition*, 109-114.

Zhang, Min, Joo Hwa Tay, Yi Qian, and Xia Sheng Gu (1997). "Comparison Between Anaerobic –Anoxic-Oxic and Anoxic-Oxic Systems for Coke Plant Wastewater Treatment", *Journal of Environmental Engineering*, 876-883.

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Run 1
no acetate

Temp	18.5
pH	7.47
ORP	-47.8
HRT	25
Q-WW	48.2
Q-MLSS	31.9
Qtot	80.1

WW1

Temp	19.6
pH	7.81
ORP	-62.8

MLSS1

Temp	19.5
pH	6.91
ORP	-12.1

WW	Average
TSS	62.5
VSS	67.5
TCOD	216
TBSCOD	77.2
RBCOD	17.6

MLSS	Average
TSS	5650
VSS	3660
TBSCOD	82.2

Influent	Average
Ortho-P	7.9
Alkalinity	175
NO3-N	15.7

Effluent	Average
TBSCOD	28.2
RBCOD	3.2
NO3-N	12.6
Ortho-P	12
Alkalinity	186

Calculated:

Infl-TSS	2287.7
Infl-VSS	1498.2
Infl-TBSCOD	79.2
Infl-RBCOD	10.6
chg NO3-N	3.1
SDNR-TC	0.124
chg TBSCOD	51
chg RBCOD	7.4
chg ALK	11
SDNR-NC	0.119

Run 2
no acetate

Temp	18.8
pH	7.54
ORP	-46.6
HRT	19
Q-WW	64
Q-MLSS	41.3
Qtot	105.3

WW1

Temp	
pH	
ORP	

Influent	Average
Ortho-P	
Alkalinity	170
NO3-N	16.1

Effluent	Average
TBSCOD	40.2
RBCOD	6.7
NO3-N	14.5
Ortho-P	9.5
Alkalinity	174

MLSS1	Average
Temp	
pH	
ORP	

Calculated:

Infl-TSS	2254
Infl-VSS	1476.5
Infl-TBSCOD	79.2
Infl-RBCOD	10.7
chg NO3-N	1.6
SDNR-TC	0.085
chg TBSCOD	39
chg RBCOD	4
chg ALK	4
SDNR-NC	0.082

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Run 3
no acetate

Temp	19.3
pH	7.52
ORP	-44
HRT	13.4
Q-WW	104.5
Q-MLSS	44.8
Qtot	149.3

WW1

Temp	
pH	
ORP	

Influent	Average
Ortho-P	25.9
Alkalinity	164
NO3-N	14.8

Effluent	Average
TBSCOD	49.2
RBCOD	10.6
NO3-N	13.9
Ortho-P	8.4
Alkalinity	166

MLSS1	Average
Temp	
pH	
ORP	

Calculated:

Infl-TSS	1739.1
Infl-VSS	1145.5
Infl-TBSCOD	78.7
Infl-RBCOD	12.3
chg NO3-N	0.9
SDNR-TC	0.086
chg TBSCOD	29.5
chg RBCOD	1.7
chg ALK	2
SDNR-NC	0.084

Run 4
with acetate

Temp	20
pH	7.33
ORP	-36.2
HRT	29.6
Q-WW	46.8
Q-MLSS	20.8
Qtot	67.6

WW1

Temp	20
pH	6.7
ORP	-5

MLSS1

Temp	20.1
pH	7.16
ORP	-29.1

WW	Average
TSS	100
VSS	40
TCOD	291
TBSCOD	132.2
RBCOD	66.7

MLSS	Average
TSS	7245
VSS	4530
TBSCOD	91.2

Influent	Average
Ortho-P	11
Alkalinity	201
NO3-N	14.1

Effluent	Average
TBSCOD	59.2
RBCOD	15.7
NO3-N	9.2
Ortho-P	
Alkalinity	219
Nitrite	

Calculated:

Infl-TSS	2298.5
Infl-VSS	1421.5
Infl-TBSCOD	119.6
Infl-RBCOD	46.2
chg NO3-N	4.9
SDNR-TC	0.168
chg TBSCOD	60.4
chg RBCOD	30.5
chg ALK	18
SDNR-NC	0.168

Run 5
with acetate

Temp	20.5
pH	7.22
ORP	-33
HRT	20.5
Q-WW	67.8
Q-MLSS	29.9
Qtot	97.7

Influent	Average
Ortho-P	32
Alkalinity	200
NO3-N	14.8

Effluent	Average
TBSCOD	73.2
RBCOD	37.4
NO3-N	11.2
Ortho-P	
Alkalinity	212

Calculated:

Infl-TSS	2286.6
Infl-VSS	1414.1
Infl-TBSCOD	119.7
Infl-RBCOD	46.3
chg NO3-N	3.6
SDNR-TC	0.176
chg TBSCOD	46.5
chg RBCOD	8.9
chg ALK	12
SDNR-NC	0.179

WW1

Temp	
pH	
ORP	

MLSS1

Temp	
pH	
ORP	

Run 6
with acetate

Temp	20.1
pH	7.18
ORP	-30.9
HRT	14.4
Q-WW	102.7
Q-MLSS	36.2
Qtot	138.9

Influent	Average
Ortho-P	25.9
Alkalinity	238
NO3-N	16.8
Nitrite	2.255

Effluent	Average
TBSCOD	78.2
RBCOD	41.1
NO3-N	14.5
Ortho-P	
Alkalinity	245

Calculated:

Infl-TSS	1962.1
Infl-VSS	1210.2
Infl-TBSCOD	121.5
Infl-RBCOD	49.3
chg NO3-N	2.3
SDNR-TC	0.19
chg TBSCOD	43.3
chg RBCOD	8.2
chg ALK	7
SDNR-NC	0.19

WW1

Temp	
pH	
ORP	

MLSS1

Temp	
pH	
ORP	

SNDR Corr Factor:	1.029

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Run 7
no acetate

Temp	17
pH	7.76
ORP	-57
HRT	38.8
Q-WW	39.5
Q-MLSS	12.1
Qtot	51.6

WW1

Temp	16.7
pH	7.72
ORP	-55.4

MLSS1

Temp	17.6
pH	6.65
ORP	3.6

Influent	Average	
TBSCOD	60.6	
RBCOD	44.8	
Ortho-P	29.1	
Alkalinity	203	
NO3-N	23	23.3
Nitrite	2.22	

Effluent	Average	
TBSCOD	12.9	
RBCOD	14.4	
NO3-N	17.9	18.1
Nitrite	3.53	
Ortho-P	24.1	
Alkalinity	220	

Calculated:

Infl-TSS	1866.65
Infl-VSS	1550
chg TBSCOD	47.7
chg RBCOD	30.4
chg ALK	17
chg NO3-N	5.15
SDNR-TC	0.134
SDNR-NC	0.123

Run 8
no acetate

Temp	17.9
pH	7.56
ORP	-46.1
HRT	14.4
Q-WW	103.1
Q-MLSS	35.4
Qtot	138.5

WW1

Temp	17.6
pH	8.19
ORP	-80.1

MLSS1	Average
Temp	18
pH	7.11
ORP	-32.7

Influent	Average	
TBSCOD	42.5	
RBCOD	30.5	
Ortho-P	29.1	
Alkalinity	210	
NO3-N	22.8	23.1
Nitrite	0.762	

Effluent	Average	
TBSCOD	22.2	
RBCOD	23.1	
NO3-N	20.5	20.8
Nitrite		
Ortho-P	23.2	
Alkalinity	216	

Calculated:

Infl-TSS	2206.665
Infl-VSS	1826.7
chg TBSCOD	20.3
chg RBCOD	7.4
chg ALK	6
chg NO3-N	2.3
SDNR-TC	0.134
SDNR-NC	0.126

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Run 9
with acetate

Temp	18.3
pH	7.84
ORP	-61.1
HRT	37.8
Q-WW	36.3
Q-MLSS	16.6
Qtot	52.9

WW1

Temp	17.8
pH	7.81
ORP	-59.6

MLSS1

Temp	18.4
pH	7.12
ORP	-20.9

Influent	Average	
TBSCOD	200	
RBCOD	153.8	
Ortho-P	32.9	
Alkalinity	221	
NO3-N	24.4	25
Nitrite	2.51	

Effluent	Average	
TBSCOD	153.6	
RBCOD	127.2	
NO3-N	16.6	16.3
Nitrite	2.93	
Ortho-P	45.8	
Alkalinity	252	

Calculated:

Infl-TSS	2375
Infl-VSS	1985
chg TBSCOD	46.4
chg RBCOD	26.6
chg ALK	31
chg NO3-N	8.25
SDNR-TC	0.166
SDNR-NC	0.158

Run 10
with acetate

Temp	19.3
pH	7.75
ORP	-56.3
HRT	15.1
Q-WW	96.8
Q-MLSS	35.5
Qtot	132.3

WW1

Temp	19.2
pH	7.73
ORP	-55.7

Influent	Average	
TBSCOD	181	
RBCOD	146	
Ortho-P	26.2	
Alkalinity	228	
NO3-N	25.3	24.9
Nitrite		

Effluent	Average	
TBSCOD	158.8	
RBCOD	141	
NO3-N	21.4	21.3
Nitrite		
Alkalinity	240	
Ortho-P	29.1	

MLSS1

Temp	19
pH	7.09
ORP	-19.5

Calculated:

Infl-TSS	2376.7
Infl-VSS	1983.4
chg TBSCOD	22.2
chg RBCOD	5
chg ALK	12
chg NO3-N	3.75
SDNR-TC	0.184
SDNR-NC	0.18

SNDR Corr Factor:	1.029

PLANT EFFLUENT

Ortho-P	12.1	
SCOD	38.4	
FCOD	36.8	30.8

Location: Olympus Terrace WWTP
Date: 5 Aug 98 & 15 Sep 98

Titrant	H_2SO_4
Normality	0.02

Alkalinity(mg $CaCO_3$/L) to pH 4.3

Runs w/o acetate

	Sample size	Start pt	End pt	Titrant used	Total ALK
Run 1 Infl	100	11.9	29.4	17.5	175
Run 2 Infl	100	26.6	43.6	17	170
Run 3 Infl	100	10.2	26.6	16.4	164
Run 7 Infl	100	20.5	40.8	20.3	203
Run 8 Infl	100	23.5	44.5	21	210
Run 1 Effl	100	1.2	19.8	18.6	186
Run 2 Effl	100	2.3	19.7	17.4	174
Run 3 Effl	100	19	36	16.6	166
Run 7 Effl	100	0	22	22	220
Run 8 Effl	100	1.9	23.5	21.6	216

Chg in ALK (mg/L)

Run 1	11
Run 2	4
Run 3	2
Run 7	17
Run 8	6

Nitrate Use (mg/L)

Run 1	3.1
Run 2	1.1
Run 3	0.6
Run 7	4.8
Run 8	1.7

Runs with Acetate

	Sample size	Start pt	End pt	Titrant used	Total ALK
Run 4 Infl	100	20	40.1	20.1	201
Run 5 Infl	100	0	20	20	200
Run 6 Infl	100	24.2	48	23.8	238
Run 9 Infl	100	25.4	47.5	22.1	221
Run 10 Infl	100	24.3	47.1	22.8	228
Run 4 Effl	100	8.2	30.1	21.9	219
Run 5 Effl	100	1.9	23.1	21.2	212
Run 6 Effl	100	11.6	36.1	24.5	245
Run 9 Effl	100	0.7	25.9	25.2	252
Run 10 Effl	100	0.1	24.1	24	240

Chg in ALK (mg/L)

Run 4	18
Run 5	12
Run 6	7
Run 9	31
Run 10	12

Nitrate Use (mg/L)

Run 4	5
Run 5	3.4
Run 6	2
Run 9	8.7
Run 10	3.4

Location: Olympus Terrace WWTP
Date: 5 August 1998

Run 1 Effl	SCOD	65
	FCOD	35.6
	RBCOD	3.2
	TBSCOD	28.2
	BSCOD	25

Run 2 Effl	SCOD	77
	FCOD	39.1
	RBCOD	6.7
	TBSCOD	40.2
	BSCOD	33.5

Run 3 Effl	SCOD	86
	FCOD	43
	RBCOD	10.6
	TBSCOD	49.2
	BSCOD	38.6

WW (Runs 1-3)	TCOD	216
	SCOD	114
	FCOD	50
	RBCOD	17.6
	TBSCOD	77.2
	BSCOD	59.6

MLSS Runs 1-3	SCOD	119
	BSCOD	82.2

Run 4 Effl	SCOD	96
	FCOD	48.1
	RBCOD	15.7
	TBSCOD	59.2
	BSCOD	43.5

Run 5 Effl	SCOD	110
	FCOD	69.8
	RBCOD	37.4
	TBSCOD	73.2
	BSCOD	35.8

Run 6 Effl	SCOD	115
	FCOD	73.5
	RBCOD	41.1
	TBSCOD	78.2
	BSCOD	37.1

WW Runs (4-6)	TCOD	291
	SCOD	169
	FCOD	99.1
	RBCOD	66.7
	TBSCOD	132.2
	BSCOD	65.5

MLSS Runs 4-6	SCOD	128
	BSCOD	91.2

Location: Olympus Terrace WWTP
Date: 15 Sep,98

Run 7 Effl	SCOD	51.3
	FCOD	45.2
	RBCOD	14.4
	TBSCOD	12.9
	BSCOD	-1.5

Run 9 Effl	SCOD	192
	FCOD	158
	RBCOD	127.2
	TBSCOD	153.6
	BSCOD	26.4

Run 7 Infl	SCOD	99
	FCOD	75.6
	RBCOD	44.8
	TBSCOD	60.6
	BSCOD	15.8

Run 9 Infl	SCOD	238.4
	FCOD	184.6
	RBCOD	153.8
	TBSCOD	200
	BSCOD	46.2

Run 8 Effl	SCOD	60.6
	FCOD	53.9
	RBCOD	23.1
	TBSCOD	22.2
	BSCOD	-0.9

Run 10 Effl	SCOD	197.2
	FCOD	171.8
	RBCOD	141
	TBSCOD	158.8
	BSCOD	17.8

Run 8 Infl	SCOD	80.9
	FCOD	61.3
	RBCOD	30.5
	TBSCOD	42.5
	BSCOD	12

Run 10 Infl	SCOD	219.4
	FCOD	176.8
	RBCOD	146
	TBSCOD	181
	BSCOD	35

PLANT EFFLUENT (25 Aug 98)

SCOD	36.8
FCOD	32.4

PLANT EFFLUENT (22 Sep 98)

SCOD	38.4
FCOD	30.8

VSS/TSS ANALYSIS

Location: Olympus Terrace WWTP
Date: **5-Aug-98**

	tare weight	105 C	550 C	TSS(mg/L)	VSS(mg/L)	Samp Sz
ML Runs 1-3	1.4127	1.4703	1.433	5760	3730	10
ML Runs 1-3	1.4069	1.4623	1.4264	5540	3590	10
ML Runs 4-6	1.4124	1.4856	1.4397	7320	4590	10
ML Runs 4-6	1.4145	1.4862	1.4415	7170	4470	10
WW Runs 1-3	1.4259	1.4271	1.4257	60	70	20
WW Runs 1-3	1.4264	1.4277	1.4264	65	65	20
WW Runs 4-6	1.4168	1.4189	1.4182	105	35	20
WW Runs 4-6	1.4183	1.4202	1.4193	95	45	20

ML Runs 1-3	
VSS (mg/L)	TSS (mg/L)
3660	5650

ML Runs 4-6	
VSS (mg/L)	TSS (mg/L)
4530	7245

VSS/TSS
Ratio
0.64

WW Runs 1-3	
VSS (mg/L)	TSS (mg/L)
67.5	62.5

WW Runs 4-6	
VSS (mg/L)	TSS (mg/L)
40	100

Location: Olympus Terrace WWTP
Date: **15-Sep-98**

	tare weight	105 C	550 C	TSS(mg/L)	VSS(mg/L)	Samp Sz
ML Run 7	1.4293	1.4569	1.434	1840	1526.7	15
ML Run 7	1.4353	1.4637	1.4401	1893.3	1573.3	15
ML Run 8	1.416	1.4495	1.4218	2233.33	1846.7	15
ML Run 8	1.4486	1.4813	1.4542	2180	1806.7	15
ML Run 9	1.4419	1.4655	1.4456	2360	1990	10
ML Run 9	1.4408	1.4647	1.4449	2390	1980	10
ML Run 10	1.4444	1.4787	1.4498	2286.7	1926.7	15
ML Run 10	1.4293	1.4663	1.4357	2466.7	2040	15

Run 7 Ave

VSS	TSS
1550	1866.65

Run 8 Ave

VSS	TSS
1826.7	2206.665

VSS/TSS
Ratio
0.83

Run 9 Ave

VSS	TSS
1985	2375

Run 10 Ave

VSS	TSS
1983.4	2376.7

TOTAL
VSS/TSS
Ratio
0.76

Location	Olympus Terrace WWTP
Date of sample	5-Aug-98

	tare weight	105 C	550 C	TSS	VSS	Samp Sz (mL)
Sample 1	1.4368	1.4689	1.4486	6420	4060	5
Sample 2	1.4402	1.4726	1.4519	6480	4140	5

MLSS characteristics
After Aeration

	Test #1	Test #2	Average
TSS	6420	6480	6450
VSS	4060	4140	4100

2.86	$g O_2 / g NO_3$	
0.8	$g O_2 / g COD$	
1.42	$g O_2 / g biomass$	

Fraction which can Denitrify

	Sample A
Time	DO
(min)	(mg/L)
0	7.59
1	6.53
2	5.8
3	5.06
4	4.14
5	3.18
6	2.29
7	1.36
8	0.46
9	0.11
10	0.09
15	0.08
20	
25	
30	

Temperature:	16.6

Temp Correction Factor:
$OUR_{20}=OUR_T/(@^{T-20})$

@=	1.029

$SEOUR_{NC}$:	0.3026
$SEOUR_{TC}$:	0.3335

Endogenous Decay Coefficient for O2 use (b):	0.23

	Sample A
Time	NO_3 Conc
(min)	(mg/L)
0	15.5
20	15.5
40	16
60	13.5
80	10
100	4.5
120	0.8

Temperature:	23.9

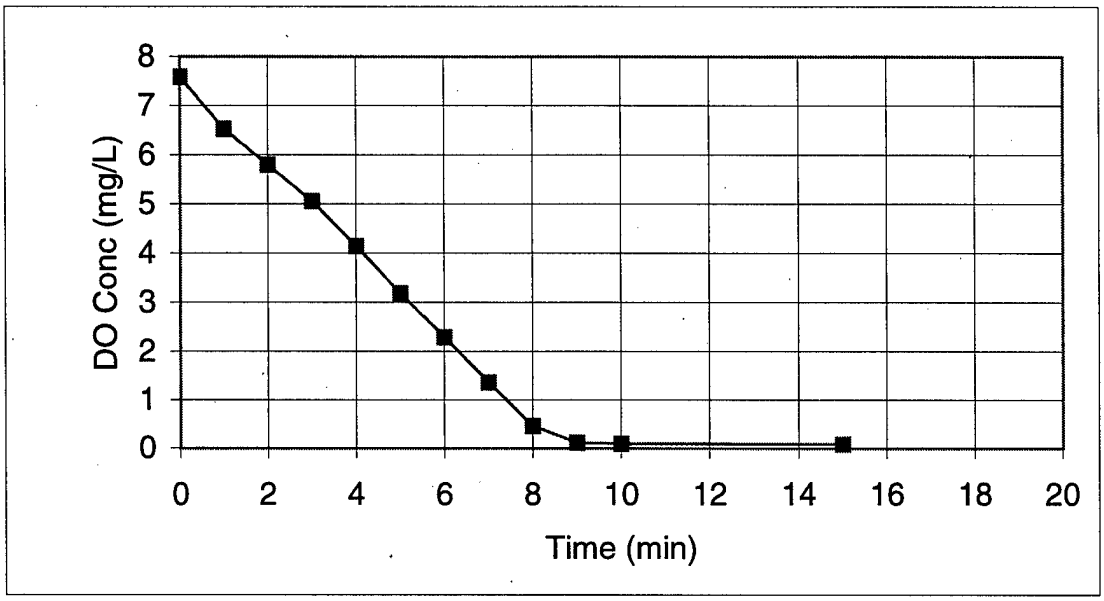
Temp Correction Factor:
$SDNR_{20}=(SDNR_T)/(@^{T-20})$

@=	1.029

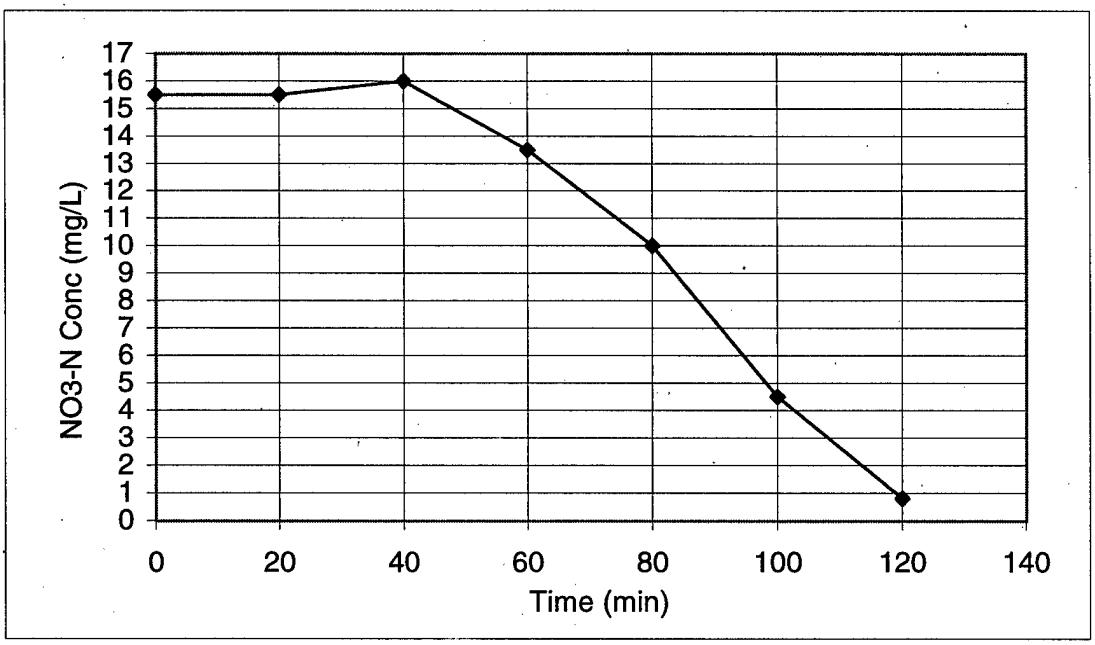
$SDNR_{NC}$:	0.0673
$SDNR_{TC}$:	0.0602

Fraction Denitrifiying:	0.516

Endogenous Decay Coefficient for NO3 use (b):	0.12



Slope: -0.87 Intercept 7.54



Slope: -0.19 Intercept: 24.3

Active Biomass Calculations

Location: Olympus Terrace WWTP (no Primary Treatment)
Data Used: August Monthly Discharge Report

Reported Data

Y (g/g-d)	
MCRT (d)	13
$CBOD_{FE}$ (mg/L)	5.1
$CBOD_{FE}$ (lb/d)	58
AB (lbs/d)	18635
WAS (lbs/d)	1547.8

V (MG)	1.1
Q (MGD)	1.35
$CBOD_{PE}$ (mg/L)	192
$CBOD_{PE}$ (lb/d)	2229
$CBOD_{PE}/COD_{PE}$	0.37
$CBOD_{FE}/COD_{FE}$	0.12

Formulas

X_T	Px*(SRT)/V
X_{bio}	Y_{net}*(chg BOD)*Q
Ynet	Y/(1+b*SRT)
Y_{obs}	Px/(chg BOD)

b	0.08	
conversion	8.34	lb/MG-mg/L
Y	0.6	g TSS/g BOD

Calculations

Y_{net}	0.29
Y_{obs}	0.71
X_T	2193.3
X_{bio}	610.25
X_{bio}/X_T	0.39

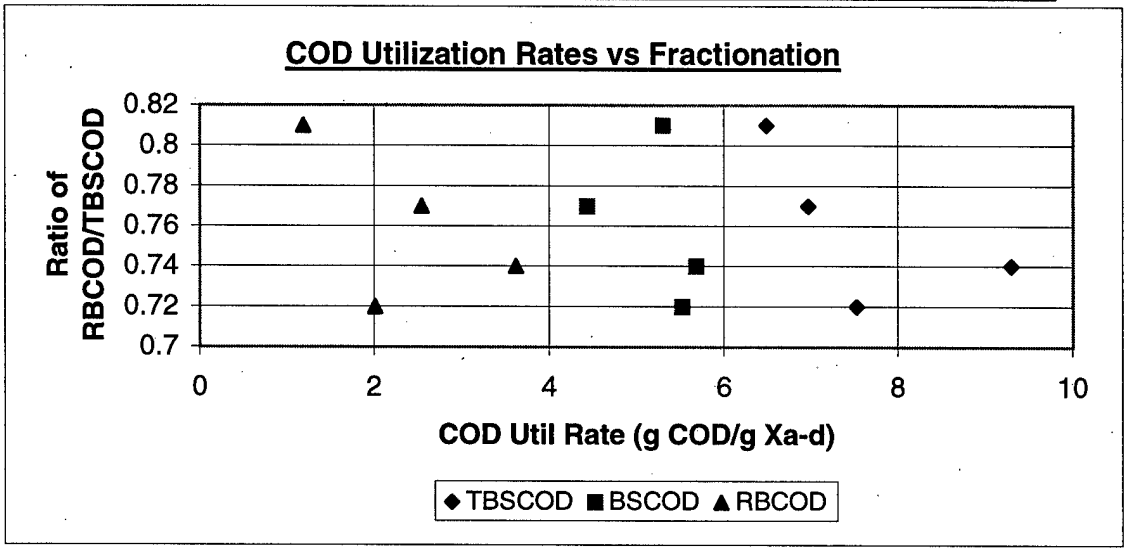
SDNR vs RBCOD
Olympus Terrace WWTP
Date: 5 Aug 98 & 15 Sep 98

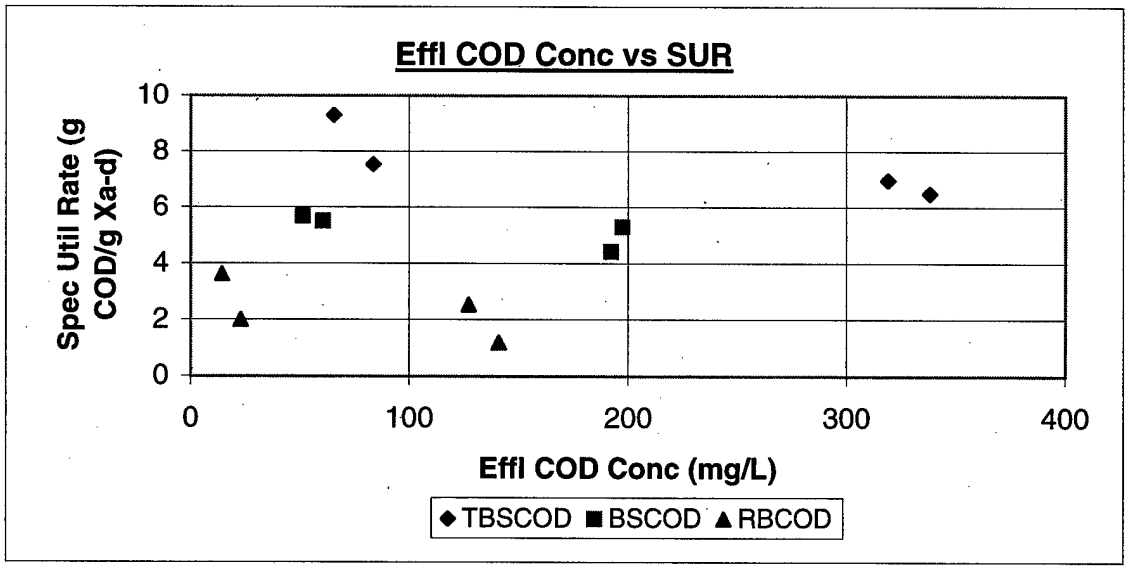
2.86	g O_2 / g NO_3
0.8	g O_2 / g COD
0.12	g $X_{d\ death}$/gX-d
1.42	g O_2 / g biomass

RUN	HRT	Temp	effl RBCOD	chg RBCOD	chg BSCOD	SDNR-NC	SDNR-TC	$SDNR_{AVLSS}$	$SDNR_{FRAVLSS}$
	(min)	(°C)	(mg/L)	(mg/L)	(mg/L)	(gNO3/gVSS-d)	(gNO3/gVSS-d)	(gNO3/gAVSS-d)	(gNO3/gAVSS-d)
Run 1	25	18.5	3.2	7.4	51	0.119	0.124	0.305	0.591
Run 2	19	18.8	6.7	4	39	0.082	0.085	0.21	0.407
Run 3	13.4	19.3	10.6	1.7	29.5	0.084	0.086	0.215	0.417
Run 4	29.6	20	15.7	30.5	60.4	0.168	0.168	0.431	0.835
Run 5	20.5	20.5	37.4	8.9	46.5	0.179	0.176	0.459	0.89
Run 6	14.4	20.1	41.1	8.2	43.3	0.19	0.19	0.487	0.944
Run 7	38.8	17	14.4	30.4	47.7	0.123	0.134	0.315	0.61
Run 8	14.4	17.9	23.1	7.4	20.3	0.126	0.134	0.323	0.626
Run 9	37.8	18.3	127.2	26.6	46.4	0.158	0.166	0.405	0.785
Run 10	15.1	19.3	141	5	22.2	0.18	0.184	0.462	0.895

RUN	HRT	VSS	effl BSCOD	effl RBCOD	chg NO3	NO3 for respiration	NO3 for for COD_{use}	R_{BSCOD}	R_{RBCOD}
	(min)	(mg/L)	(mg/L)	(mg/L)	(mg/L)	(mg/L)	(mg/L)	(gBSCOD/gXa-d)	(gRBCOD/gXa-d)
Run 1	25	1498	65	3.2	3.1	1.55	1.55	9.74	1.41
Run 2	19	1477	77	6.7	1.6	1.16	0.44	9.95	1.02
Run 3	13.4	1146	86	10.6	0.9	0.64	0.26	13.75	0.79
Run 4	29.6	1422	96	15.7	4.9	1.74	3.16	10.27	5.19
Run 5	20.5	1414	110	37.4	3.6	1.2	2.4	11.48	2.2
Run 6	14.4	1210	115	41.1	2.3	0.72	1.58	17.78	3.37
Run 7	38.8	1550	51.3	14.4	5.15	2.49	2.66	5.68	3.62
Run 8	14.4	1827	60.6	23.1	2.3	1.09	1.21	5.52	2.01
Run 9	37.8	1985	192	127.2	8.25	3.1	5.15	4.43	2.54
Run 10	15.1	1983	197.2	141	3.75	1.24	2.51	5.3	1.19

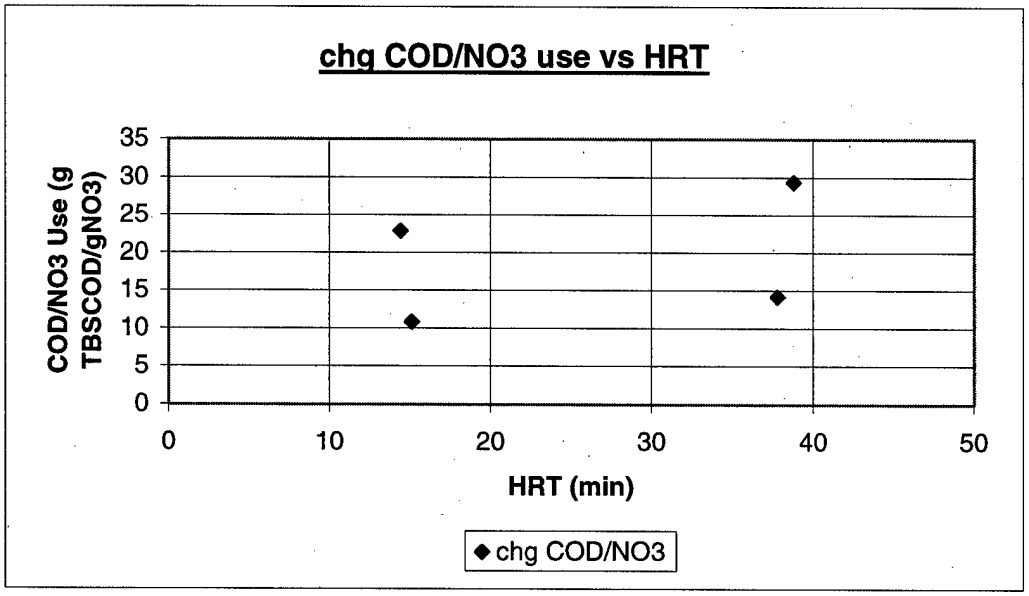
Run	effl TBSCOD (mg/L)	effl BSCOD (mg/L)	effl RBCOD (mg/L)	R_{TBSCOD} (gCOD/gXa-d)	R_{BSCOD} (gCOD/gXa-d)	R_{RBCOD} (gCOD/gXa-d)	Infl RBCOD/ Infl TBSCOD (g/g)
Run 7	65.7	51.3	14.4	9.3	5.68	3.62	0.74
Run 8	83.7	60.6	23.1	7.53	5.52	2.01	0.72
Run 9	319.2	192	127.2	6.97	4.43	2.54	0.77
Run 10	338.2	197.2	141	6.49	5.3	1.19	0.81

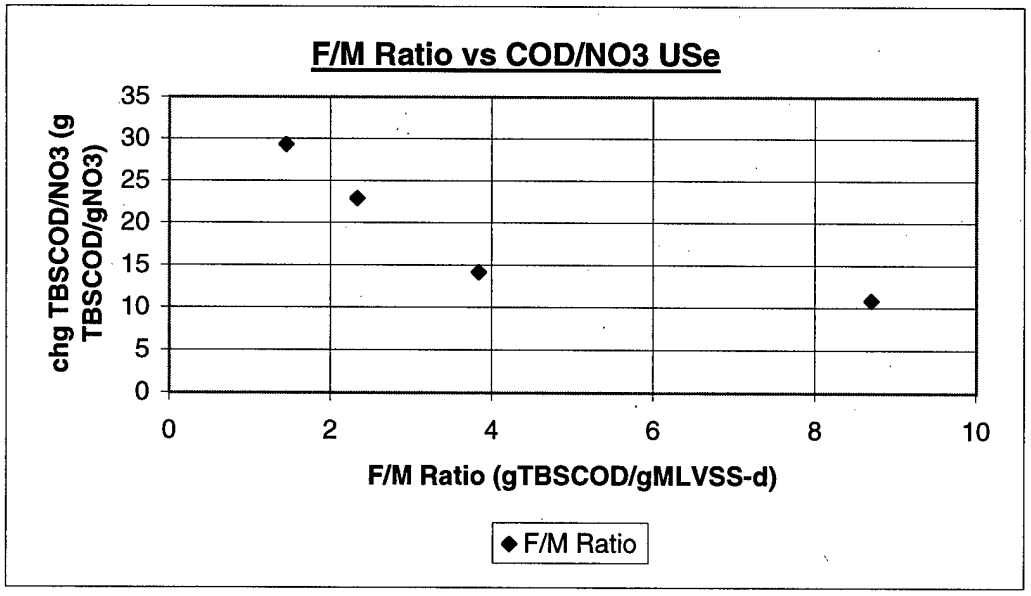




Olympic Terrace Experiments

RUN	chg TBSCOD	NO_3 for COD_{use}	chg COD/NO_3 (mg TBSCOD /mg NO3)	F/M Ratio (g TBSCOD /g MLVSS-d)	HRT (min)
	(mg TBSCOD/L)	(mg/L)			
Run 7	78.1	2.66	29.36	1.45	38.8
Run 8	27.7	1.21	22.89	2.33	14.4
Run 9	73	5.15	14.17	3.84	37.8
Run 10	27.2	2.51	10.84	8.7	15.1





Plant Specific General Operations Information

Location: Olympic Terrace WWTP, Muckleteo, WA

Parameter	*July Average*	*Aug Average*	*Units*
MLSS	2217	2031	mg/L
MLVSS	1866	1717	mg/L
Daily Flow	1.45	1.35	MGD
MCRT	13	13	d^{-1}
F/M Ratio (COD)	0.16	0.18	
Hetertrophic Yield	n.a.	n.a.	g/g
MLSS SVI	137	150	
WWSS	222	179	mg/L
WWVSS	n.a.	n.a.	mg/L
Raw Infl TBOD conc	222	245	mg/L
Raw Infl COD conc	499	513	mg/L
Raw Infl NH_3	n.a.	n.a.	mg/L
Raw Infl NO_3	n.a.	n.a.	mg/L
Raw Infl NO_2	n.a.	n.a.	mg/L
Raw Infl pH	7.77	7.79	
Raw Infl DO	5.7	5.3	mg/L
Raw Infl Temp	19.4	20.5	^{o} C
Raw Infl Alk	n.a.	n.a.	mg/L (as $CaCO_3$)
Final Effl COD	42.5	31	mg/L(% removal)
Final Effl NH_3	1.9	2.3	mg/L
Final Effl NO_3	n.a.	n.a.	mg/L
Final Effl NO_2	n.a.	n.a.	mg/L
Final Effl PO_4	n.a.	n.a.	mg/L
Final Effl ALK	n.a.	n.a.	mg/L (as $CaCO_3$)
Final Effl CBOD	5.8, 73	5.1, 2229	mg/L, lbs/d
Infl CBOD	179, 2240	1,922,229	mg/L, lbs/d
AB LB	n.a.	18635	lbs/d
Sludge Production	1644.4	1547.6	lbs/d

Analytical Report (Snoqualmie Falls) 17 August 1998

Run #1
no acetate

Temp	24
pH	7.32
ORP	-37.9
HRT	30.2
Q-WW	45.6
Q-MLSS	20.6
Qtot	66.2

WW1

Temp	22.5
pH	7.28
ORP	-38.4

MLSS1

Temp	24
pH	6.91
ORP	-17

WW	Average
TSS	72.5
VSS	58.75
TCOD	188
TBSCOD	59.6
RBCOD	11.6

MLSS	Average
TSS	5660
VSS	4650
TBSCOD	83

Influent	Average
Ortho-P	11
Alkalinity	167
NO3-N	14.5
Nitrite	0.0812

Effluent	Average
TBSCOD	-1.8
RBCOD	-1.3
NO3-N	13.5
Nitrite	0.0078
Ortho-P	11.2
Alkalinity	171

Calculated:

Infl TSS	1811.2
Infl VSS	1487.4
Infl TBSCOD	66.9
Infl RBCOD	8
chg NO3-N	1
SDNR-CT	0.029
SDNR-NC	0.032
chg TBSCOD	68.7
chg RBCOD	9.3
chg ALK	4

Run #2
no acetate

Temp	25
pH	7.23
ORP	-34.8
HRT	20.7
Q-WW	64
Q-MLSS	32.6
Qtot	96.6

WW1

Temp	23
pH	7.31
ORP	-39.1

Influent	Average
Ortho-P	12.8
Alkalinity	168
NO3-N	14
Nitrite	

Effluent	Average
TBSCOD	1.9
RBCOD	-0.2
NO3-N	13.1
Ortho-P	12.8
Alkalinity	171

MLSS1	Average
Temp	24
pH	6.95
ORP	-18.6

Calculated:

Infl TSS	1958.1
Infl VSS	1608.2
Infl TBSCOD	67.5
Infl RBCOD	7.7
chg NO3-N	0.9
SDNR-CT	0.034
SDNR-NC	0.039
chg TBSCOD	65.6
chg RBCOD	7.9
chg ALK	3

Analytical Report (Snoqualmie Falls) 17 August 1998

Run #3
no acetate

Temp	25.5
pH	7.13
ORP	-29.4
HRT	14.3
Q-WW	93.1
Q-MLSS	47.1
Qtot	140.2

WW1

Temp	24
pH	7.33
ORP	-40.5

Influent	Average
Ortho-P	14.4
Alkalinity	170
NO3-N	14.5

Effluent	Average
TBSCOD	7.3
RBCOD	0.9
NO3-N	13.8
Ortho-P	14.2
Alkalinity	172

MLSS1

	Average
Temp	24
pH	6.97
ORP	-20.5

Calculated:

Infl TSS	1949.6
Infl VSS	1601.2
Infl TBSCOD	67.5
Infl RBCOD	7.7
chg NO3-N	0.7
SDNR-CT	0.038
SDNR-NC	0.044
chg TBSCOD	60.2
chg RBCOD	6.8
chg ALK	2

Run #4
with acetate

Temp	26
pH	7.19
ORP	-39.4
HRT	29.5
Q-WW	44.6
Q-MLSS	23.1
Qtot	67.7

WW1

Temp	24
pH	7.28
ORP	-37.6

MLSS1

Temp	24
pH	6.8
ORP	-11.2

WW	Average
TSS	71.25
VSS	60
TCOD	204
TBSCOD	95.6
RBCOD	35.4

MLSS	Average
TSS	6190
VSS	4980
SCOD	84.1

Influent	Average
Ortho-P	10.1
Alkalinity	190
NO3-N	14.8
Nitrite	0.3848

Effluent	Average
TBSCOD	23.6
RBCOD	14.9
NO3-N	12.3
Ortho-P	12.2
Alkalinity	199
Nitrite	0.357

Calculated:

Infl TSS	2159
Infl VSS	1738.8
Infl TBSCOD	91.7
Infl RBCOD	23.3
chg NO3-N	2.5
SDNR-CT	0.059
SDNR-NC	0.07
chg TSCOD	68.1
chg RBCOD	8.4
chg ALK	9

Run #5
with acetate

Temp	26
pH	7.21
ORP	-36
HRT	21.4
Q-WW	60.6
Q-MLSS	33
Qtot	93.6

Influent	Average
Ortho-P	10
Alkalinity	192
NO3-N	14.4
Nitrite	0.1237

Effluent	Average
TBSCOD	32.8
RBCOD	22.9
NO3-N	12.9
Nitrite	0.4475
Ortho-P	10.7
Alkalinity	198

Calculated:

Infl TSS	2228.5
Infl VSS	1794.6
Infl TBSCOD	91.5
Infl RBCOD	22.9
chg NO3-N	1.5
SDNR-CT	0.047
SDNR-NC	0.056
chg TBSCOD	58.7
chg RBCOD	0
chg ALK	6

WW1

Temp	25
pH	7.31
ORP	-39.4

MLSS1

Temp	24.5
pH	6.89
ORP	-16.1

Run #6
with acetate

Temp	26.5
pH	7.3
ORP	-36.4
HRT	14.1
Q-WW	94.6
Q-MLSS	47.4
Qtot	142

Influent	Average
Ortho-P	9.2
Alkalinity	187
NO3-N	14.2

Effluent	Average
TBSCOD	39.3
RBCOD	23.4
NO3-N	12.6
Ortho-P	10.3
Alkalinity	192

Calculated:

Infl TSS	2113.7
Infl VSS	1702.3
Infl TBSCOD	91.8
Infl RBCOD	23.6
chg NO3-N	1.6
SDNR-CT	0.08
SDNR-NC	0.096
chg TBSCOD	52.5
chg RBCOD	0.2
chg ALK	5

WW1

Temp	25
pH	7.5
ORP	-45.1

MLSS1

Temp	25
pH	7.1
ORP	-26.3

SNDR Corr Factor:	1.029

Location: Snoqualime Falls
Date: 17-Aug-98

ALKALINITY (mg $CaCO_3$/L) to pH 4.3

Titrant	H_2SO_4
Normality	0.02

	Sample size	Start pt	End pt	Titrant used	Total ALK
Run 1 Infl	100	0.6	17.3	16.7	167
Run 2 Infl	100	17.3	34.1	16.8	168
Run 3 Infl	100	1.3	18.3	17	170
Run 1 Effl	100	18.2	35.3	17.1	171
Run 2 Effl	100	0.7	17.8	17.1	171
Run 3 Effl	100	17.8	35	17.2	172
Run 4 Infl	100	0.9	19.9	19	190
Run 5 Infl	100	19.9	39.1	19.2	192
Run 6 Infl	100	3.7	22.4	18.7	187
Run 4 Effl	100	6.3	26.2	19.9	199
Run 5 Effl	100	1.2	21	19.8	198
Run 6 Effl	100	21	40.2	19.2	192

Chg in ALK (mg/L)

Run 1	4
Run 2	3
Run 3	2
Run 4	9
Run 5	6
Run 6	5

Nitrate Use (mg/L)

Run 1	1.1
Run 2	0.8
Run 3	0.6
Run 4	2.5
Run 5	1.7
Run 6	1.4

Location: Snoqualmie Falls
Date: 17-Aug-98

Run 1	SCOD	28.6
	FCOD	27.5
	RBCOD	-1.3
	TBSCOD	-1.8
	BSCOD	-3.1

Run 2	SCOD	32.3
	FCOD	28.6
	RBCOD	-0.2
	TBSCOD	1.9
	BSCOD	1.7

Run 3	SCOD	37.7
	FCOD	29.7
	RBCOD	0.9
	TBSCOD	7.3
	BSCOD	6.4

WW1	TCOD	188
	SCOD	90
	FCOD	40.4
	RBCOD	11.6
	TBSCOD	59.6
	BSCOD	48

MLSS	SCOD	113.4
	TBSCOD	83

Run 4	SCOD	54
	FCOD	43.7
	RBCOD	14.9
	TBSCOD	23.6
	BSCOD	8.7

Run 5	SCOD	63.2
	FCOD	51.7
	RBCOD	22.9
	TBSCOD	32.8
	BSCOD	9.9

Run 6	SCOD	69.7
	FCOD	52.2
	RBCOD	23.4
	TBSCOD	39.3
	BSCOD	15.9

WW2	TCOD	204
	SCOD	126
	FCOD	64.2
	RBCOD	35.4
	TBSCOD	95.6
	BSCOD	60.2

MLSS	SCOD	84.1
	TBSCOD	53.7

PLANT EFFLUENT

FCOD	28.8
SCOD	30.4

VSS/TSS Analysis

	tare weight	105 C	550 C	TSS	VSS	Samp Sz	VSS/TSS
MLSS1A	1.4283	1.4567	1.4334	5680	4660	5	0.82
MLSS1B	1.4217	1.4499	1.4267	5640	4640	5	0.82
MLSS2A	1.4223	1.4534	1.4284	6220	5000	5	0.8
MLSS2B	1.4193	1.4501	1.4253	6160	4960	5	0.81
WW1A	1.4137	1.4165	1.4142	70	57.5	40	0.82
WW1B	1.4254	1.4284	1.426	75	60	40	0.8
WW2A	1.4397	1.4427	1.4402	75	62.5	40	0.83
WW2B	1.4307	1.4334	1.4311	67.5	57.5	40	0.85

MLSS1 Ave

VSS	TSS
4650	5660

WW1 Ave

VSS	TSS
58.75	72.5

MLSS2 Ave

VSS	TSS
4980	6190

WW2 Ave

VSS	TSS
60	71.25

ML AVE
VSS/TSS
0.81

MLSS ANALYSIS

Location	Snoqualmie Falls
Date of sample	17-Aug-98

	tare weight	105 C	550 C	TSS	VSS	Samp Sz (mL)
MLSS-OURA	1.4329	1.4608	1.4375	5580	4660	5
MLSS-OURB	1.4296	1.4603	1.4342	6140	5220	5

MLSS characteristics
After Aeration

	Test #1	Test #2	Average
TSS	5580	6140	5860
VSS	4660	5220	4940

2.86	g O_2 / g NO_3
0.8	g O_2 / g COD
1.42	g O_2 / g biomass

Fraction which can Denitrify

	Sample A
Time	DO Conc
(min)	(mg/L)
0	7.92
1	7.34
2	6.85
3	6.3
4	5.74
5	5.21
6	4.65
7	4.09
8	3.56
9	3.01
10	2.45
15	0.13
20	0.1
25	
30	
35	

Temperature:	18.2

Temp Correction Factor:
$OUR_{20}=OUR_T/(@^{T-20})$

@=	1.029

$SEOUR_{NC}$:	0.1584
$SEOUR_{TC}$:	0.1667

Endogenous Decay Coefficient for O2 use (b):	0.12

	Sample A
Time	NO_3 Conc
(min)	(mg/L)
0	14.2
30	13.9
50	12.9
70	9.7
90	7.1
110	3.9
130	1.3

Temperature:	24.1

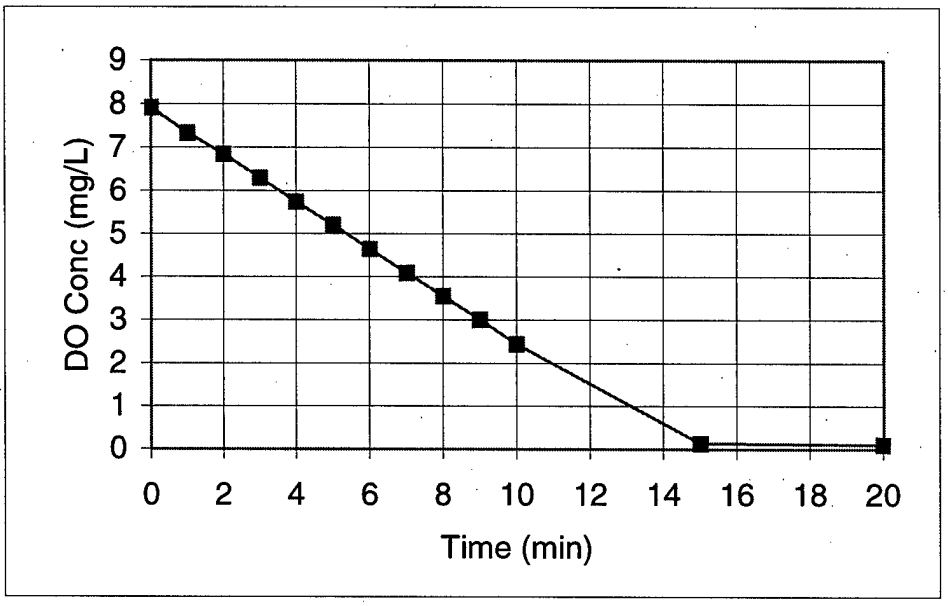
Temp Correction Factor:
$SDNR_{20}=(SDNR_T)/(@^{T-20})$

@=	1.029

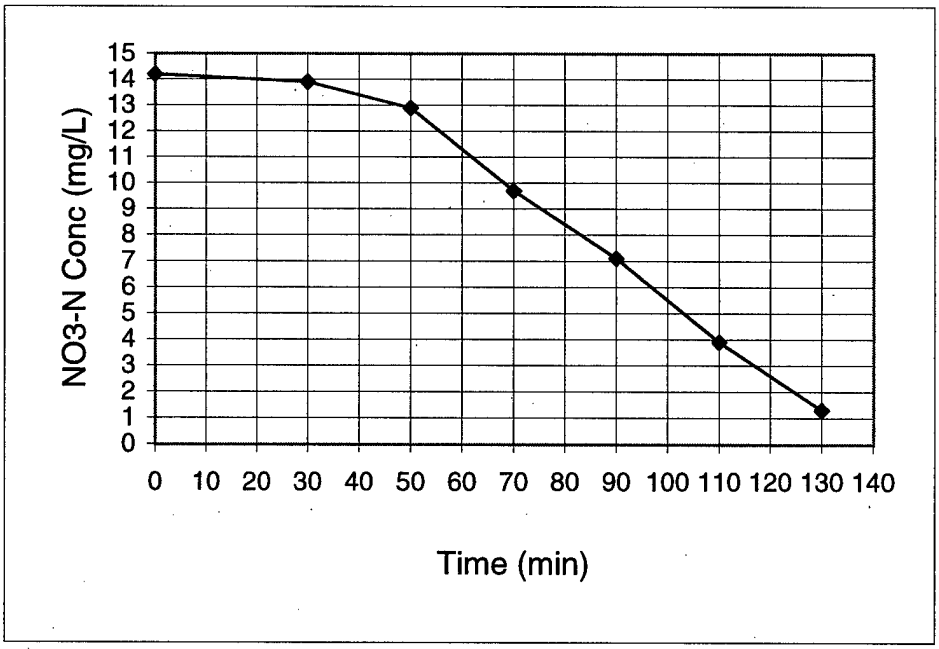
$SDNR_{NC}$:	0.0437
$SDNR_{TC}$:	0.0389

Fraction Denitrifiy	0.667

Endogenous Decay Coefficient for NO3 use (b):	0.08



Slope: -0.55 Intercep 7.92



Slope: -0.13 Intercep 18.53

Active Biomass Calculations

Location: Snoqualmie Falls WWTP (no Primary Treatment)
Data Used: August Monthly Discharge Report

Reported Data

Y (g/g-d)	
MCRT (d)	23
$CBOD_{FE}$ (mg/L	1.5
$CBOD_{FE}$ (lb/d)	3.65
AB (lb/d)	6548
WAS (lb/d)	284.7

V (MG)	1.1
Q (MGD)	0.292
$CBOD_{PE}$ (mg/L	76.25
$CBOD_{PE}$ (lb/d)	185.7
$CBOD_{PE}/COD_F$	0.37
$CBOD_{FE}/COD_F$	0.12

Formulas

X_T	Px*(SRT)/V
Px_{bio}	Y_{net}*(chg BOD)*Q
Ynet	Y/(1+b*SRT)
Y_{obs}	Px/(chg BOD)

b	0.08	
conversion	8.34	lb/MG-mg/L
Y	0.6	g TSS/g BOD

Calculations

Y_{net}	0.21
Y_{obs}	0.64
$X_{T\ (mg/L)}$	713.8
$Px_{bio\ (lb/d)}$	38.23
X_{bio}/X_T	0.13

SDNR vs FCOD
Snoqualmie Falls

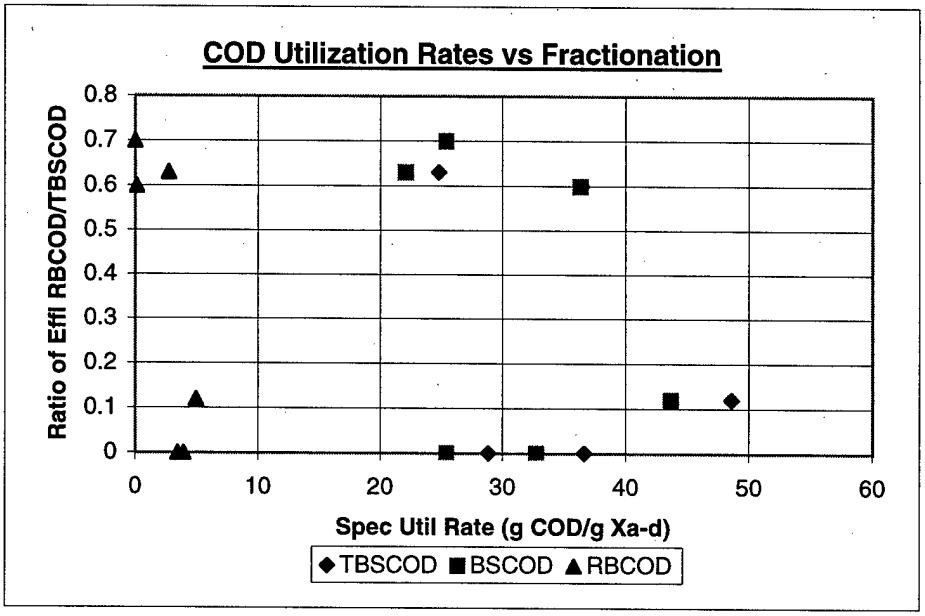
Theortical AVSS
0.3

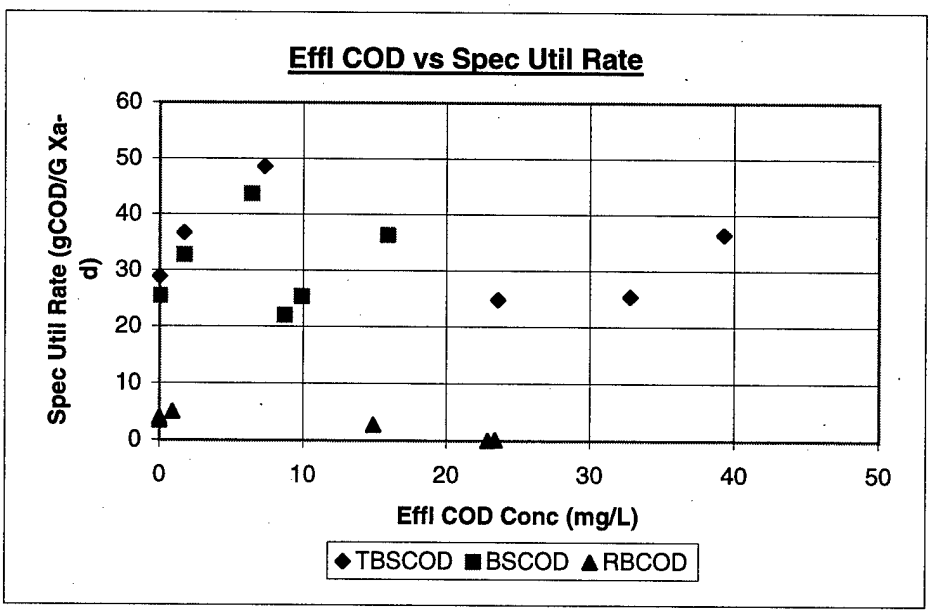
2.86	g O_2 / g NO_3
0.8	g O_2 / g COD
0.08	g $X_{d\ death}$/gX-d
1.42	g O_2 / g biomass

RUN	HRT	Temp	effl RBCOD	chg RBCOD	chg BSCOD	SDNR-NC	SDNR-TC	$SDNR_{AVLSS}$	$SDNR_{FRAVLSS}$
	(min)	(°C)	(mg/L)	(mg/L)	(mg/L)	(gNO3/gVSS-d)	(gNO3/gVSS-d)	(gNO3/gAVSS-d)	(gNO3/gAVSS-d)
1	30.2	24	0	9.3	68.7	0.032	0.029	0.097	0.145
2	20.7	25	0	7.9	65.6	0.039	0.034	0.113	0.169
3	14.3	25.5	0.9	6.8	60.2	0.044	0.038	0.127	0.19
4	29.5	26	14.9	8.4	68.1	0.07	0.059	0.197	0.295
5	21.4	26	22.9	0	58.7	0.056	0.047	0.157	0.235
6	14.1	26.5	23.4	0.2	52.5	0.096	0.08	0.267	0.4

RUN	HRT	VSS	effl BSCOD	effl RBCOD	chg NO3	NO3 for respiration	NO3 for COD_{use}	R_{BSCOD}	R_{RBCOD}
	(min)	(mg/L	(mg/L)	(mg/L)	(mg/L)	(mg/L)	(mg/L)	(gBSCOD/gXa-d)	gRBCOD/gXa-d)
1	30.2	1487	0	0	1	1.24	-0.24	25.4	3.44
2	20.7	1608	1.7	0	0.9	0.92	-0.02	32.73	3.94
3	14.3	1601	6.4	0.9	0.7	0.63	0.07	43.66	4.93
4	29.5	1739	8.7	14.9	2.5	1.41	1.09	22.05	2.72
5	21.4	1795	9.9	22.9	1.5	1.06	0.44	25.38	0
6	14.1	1702	15.9	23.4	1.6	0.66	0.94	36.32	0.14

Run	effl TBSCOD	effl BSCOD	effl RBCOD	R_{TBSCOD}	R_{BSCOD}	R_{RBCOD}	R_{RBCOD}/R_{TBSCOD}
	(mg/L)	(mg/L)	(mg/L)	(gCOD/gXa-d)	(gCOD/gXa-d)	(gCOD/gXa-d)	(g/g)
1	0	0	0	28.84	25.4	3.44	#DIV/0!
2	1.7	1.7	0	36.67	32.73	3.94	0
3	7.3	6.4	0.9	48.59	43.66	4.93	0.12
4	23.6	8.7	14.9	24.77	22.05	2.72	0.63
5	32.8	9.9	22.9	25.38	25.38	0	0.7
6	39.3	15.9	23.4	36.46	36.32	0.14	0.6





Location: Snoqualmie Falls WWTP, Snoqualmie, WA

Parameter	*July Average*	*Aug Average*	*Units*
MLSS	n.a.	2217	mg/L
MLVSS	n.a.	n.a.	mg/L
Daily Flow	0.277	0.292	MGD
MCRT	40	23	d^{-1}
F/M Ratio (COD)	n.a.	n.a.	
Hetertrophic Yield	n.a.	n.a.	g/g
MLSS SVI	n.a.	n.a.	
WWSS	73.8	85.75	mg/L
WWVSS	n.a.	n.a.	mg/L
Raw Infl TBOD conc	n.a.	n.a.	mg/L
Raw Infl COD conc	n.a.	n.a.	mg/L
Raw Infl NH_3	n.a.	n.a.	mg/L
Raw Infl NO_3	n.a.	n.a.	mg/L
Raw Infl NO_2	n.a.	n.a.	mg/L
Raw Infl pH	7.27	7.14	
Raw Infl DO	2.51	2.64	mg/L
Raw Infl Temp	21.4	22.3	$^{\circ}C$
Raw Infl Alk	n.a.	n.a.	mg/L (as $CaCO_3$)
Final Effl COD	n.a.	n.a.	mg/L
Final Effl NH_3	n.a.	n.a.	mg/L
Final Effl NO_3	n.a.	n.a.	mg/L
Final Effl NO_2	n.a.	n.a.	mg/L
Final Effl PO_4	n.a.	n.a.	mg/L
Final Effl ALK	n.a.	n.a.	mg/L (as $CaCO_3$)
Final Effl CBOD	1.4, 3	1.5, 3.25	mg/L, lbs/d
Infl CBOD	50.2, 127.8	76.25, 150.3	mg/L, lbs/d
AB LB	7943	6548	lbs/d
Sludge Production	207	284.7	lbs/d

Analytical Report (Chambers Creek) 20 August 1998

Run 1
no acetate

WW	Average
TSS	28.3
VSS	20
TCOD	216
TBSCOD	41.4
RBCOD	29.1

Temp	27
pH	7.32
ORP	-39.9
HRT	30.2
Q-WW	45.6
Q-MLSS	20.6
Qtot	66.2

MLSS	Average
TSS	5480
VSS	3995
SCOD	209.4

WW1

Temp	22.5
pH	7.28
ORP	-38.4

Influent	Average
Ortho-P	34
Alkalinity	192
NO3-N-N	15.3

Calculated:

Infl-TSS	1724.8
Infl-VSS	1256.9
Infl-TBSCOD	93.7
Infl-RBCOD	20
chg NO3-N	2.2
SDNR-TC	0.068
chg TBSCOD	68.5
chg RBCOD	14.1
chg ALK	7.3
SDNR-NC	0.083

MLSS1

Temp	23
pH	6.91
ORP	-17

Effluent	Average
TBSCOD	25.2
RBCOD	5.9
NO3-N-N	13.1
Ortho-P	34
Alkalinity	199.3

Run 2
no acetate

Influent	Average
Ortho-P	
Alkalinity	193.3
NO3-N-N	14.4

Calculated:

Infl-TSS	1868.1
Infl-VSS	1361.5
Infl-TBSCOD	98.1
Infl-RBCOD	19.3
chg NO3-N	1.3
SDNR-TC	0.055
chg TBSCOD	44.5
chg RBCOD	8.7
chg ALK	4
SDNR-NC	0.066

Temp	26.8
pH	7.23
ORP	-34.3
HRT	20.7
Q-WW	64
Q-MLSS	32.6
Qtot	96.6

Effluent	Average
TBSCOD	53.6
RBCOD	10.6
NO3-N-N	13.1
Ortho-P	
Alkalinity	197.3

WW1

Temp	23
pH	7.31
ORP	-39.1

MLSS1	Average
Temp	23.5
pH	6.95
ORP	-18.6

Run 3
no acetate

Temp	27
pH	7.13
ORP	-29.4
HRT	14.7
Q-WW	86.6
Q-MLSS	49.8
Qtot	136.4

WW1

Temp	22.5
pH	7.33
ORP	-40.5

Influent	Average
Ortho-P	
Alkalinity	181.5
NO3-N-N	13.7

Effluent	Average
TBSCOD	55.5
RBCOD	16.1
NO3-N-N	12.7
Ortho-P	
Alkalinity	184.5

MLSS1	Average
Temp	23
pH	6.97
ORP	-20.5

Calculated:

Infl-TSS	2018.7
Infl-VSS	1471.3
Infl-TBSCOD	102.7
Infl-RBCOD	18.5
chg NO3-N	1
SDNR-TC	0.055
chg TBSCOD	47.2
chg RBCOD	2.4
chg ALK	3
SDNR-NC	0.067

Run 4
with acetate

Temp	25.2
pH	7.19
ORP	-39.4
HRT	29.5
Q-WW	44.6
Q-MLSS	23.1
Qtot	67.7

WW1

Temp	22
pH	7.28
ORP	-37.6

MLSS1

Temp	23.8
pH	6.8
ORP	-11.2

WW	Average
TSS	28.3
VSS	16.7
TCOD	279
TBSCOD	109
RBCOD	77

MLSS	Average
TSS	4845
VSS	3890
TBSCOD	211.4

Influent	Average
Ortho-P	
Alkalinity	201.5
NO3-N-N	15.3

Effluent	Average
TBSCOD	42.6
RBCOD	19.3
NO3-N-N	11.8
Ortho-P	36
Alkalinity	213.5

Calculated:

Infl-TSS	1671.8
Infl-VSS	1338.3
Infl-TBSCOD	143.9
Infl-RBCOD	50.7
chg NO3-N	3.5
SDNR-TC	0.11
chg TBSCOD	101.3
chg RBCOD	31.4
chg ALK	12
SDNR-NC	0.128

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Run 5
with acetate

Temp	25
pH	7.21
ORP	-37.4
HRT	21.4
Q-WW	60.6
Q-MLSS	33
Qtot	93.6

Influent	Average
Ortho-P	
Alkalinity	202.5
NO3-N-N	14.9

Effluent	Average
TBSCOD	63.6
RBCOD	30
NO3-N-N	12.2
Ortho-P	
Alkalinity	212.5

Calculated:

Infl-TSS	1726.5
Infl-VSS	1382.3
Infl-TBSCOD	145.1
Infl-RBCOD	49.9
chg NO3-N	2.7
SDNR-TC	0.114
chg TBSCOD	81.5
chg RBCOD	19.9
chg ALK	10
SDNR-NC	0.131

WW1

Temp	23.2
pH	7.31
ORP	-39.4

MLSS1

Temp	24
pH	6.89
ORP	-16.1

Run 6
with acetate

Temp	24.6
pH	7.3
ORP	-34
HRT	14.1
Q-WW	94.6
Q-MLSS	47.4
Qtot	142

Influent	Average
Ortho-P	
Alkalinity	202
NO3-N-N	15

Effluent	Average
TBSCOD	66.7
RBCOD	35.6
NO3-N-N	13.7
Ortho-P	
Alkalinity	205.5

Calculated:

Infl-TSS	1636.1
Infl-VSS	1309.6
Infl-TBSCOD	143.2
Infl-RBCOD	51.3
chg NO3-N	1.3
SDNR-TC	0.089
chg TBSCOD	76.5
chg RBCOD	15.7
chg ALK	3.5
SDNR-NC	0.101

WW1

Temp	23.5
pH	7.5
ORP	-49.1

MLSS1

Temp	24
pH	7.1
ORP	-26.3

SNDR Corr Factor:	1.029

Run 7
no acetate

Temp	18
pH	7.82
ORP	-55.1
HRT	62.5
Q-WW	22.5
Q-MLSS	9.5
Qtot	32

WW1

Temp	18.2
pH	7.55
ORP	-40.8

MLSS1

Temp	18.5
pH	7.08
ORP	-17

Influent	Average	
TBSCOD	26.7	
RBCOD	14.8	
Ortho-P	29.1	
Alkalinity	123	
NO3-N	21.8	21.1
Nitrite	0.762	

Effluent	Average	
TBSCOD	14.1	
RBCOD	11.9	
NO3-N	16.9	16.8
Nitrite	0.818	
Ortho-P	30.4	
Alkalinity	137	

Calculated:

Infl-TSS	1903.35
Infl-VSS	1673.3
chg TBSCOD	12.6
chg RBCOD	2.9
chg ALK	14
chg NO3-N	4.6
SDNR-TC	0.067
SDNR_NC	0.063

Run 8
with acetate

Temp	18.8
pH	8.08
ORP	-68.9
HRT	62.5
Q-WW	22.5
Q-MLSS	9.5
Qtot	32

WW1

Temp	18.5
pH	7.67
ORP	-47.3

MLSS1

Temp	18.7
pH	7.22
ORP	-23.5

Influent	Average	
TBSCOD	65.7	
RBCOD	48.5	
Ortho-P	31.3	
Alkalinity	143	
NO3-N	22.1	21.7
Nitrite	1.432	

Effluent	Average	
TBSCOD	24.5	
RBCOD	25.8	
NO3-N	13.2	13.1
Nitrite	1.706	
Ortho-P	32.6	
Alkalinity	172	

Calculated:

Infl-TSS	2160
Infl-VSS	1800
chg TBSCOD	41.2
chg RBCOD	22.7
chg ALK	29
chg NO3-N	8.75
SDNR-TC	0.116
SDNR-NC	0.112

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Run 9
with acetate

Temp	19
pH	7.75
ORP	-52.3
HRT	14.4
Q-WW	100
Q-MLSS	39.3
Qtot	139.3

Influent	Average	
TBSCOD	78.5	
RBCOD	63.9	
Ortho-P	30.4	
Alkalinity	152	
NO3-N	20.3	20.9
Nitrite		

Effluent	Average	
TBSCOD	62	
RBCOD	50.2	
NO3-N	18.5	18.7
Nitrite		
Alkalinity	159	
Ortho-P	31.2	

Calculated:

Infl-TSS	2113.3
Infl-VSS	1826.7
chg TBSCOD	16.5
chg RBCOD	13.7
chg ALK	7
chg NO3-N	2
SDNR-TC	0.113
SDNR-NC	0.109

WW1

Temp	18.8
pH	7.69
ORP	-48.4

MLSS1

Temp	18.8
pH	7.31
ORP	-28.3

SNDR Corr Factor:	1.029

PLANT EFFLUENT

Ortho-P	10.4	
SCOD	24.6	
FCOD	22.7	22.7

Location: Chamber's Creek
Date: 20 Aug 98 & 24 Sep 98

Alkalinity(mg $CaCO_3$/L) to pH 4.3

Titrant	H_2SO_4
Normality	0.02

Runs w/o acetate

	Sample size	Start pt	End pt	Titrant used	Total ALK
Run 1 Infl	150	0.8	29.6	28.8	192
Run 2 Infl	150	17.5	46.5	29	193.3
Run 3 Infl	200	1.2	37.5	36.3	181.5
Run 7 Infl	100	15.6	27.9	12.3	123
Run 1 Effl	150	9.6	39.5	29.9	199.3
Run 2 Effl	150	3.1	32.7	29.6	197.3
Run 3 Effl	200	8.7	45.6	36.9	184.5
Run 7 Effl	100	27.9	41.6	13.7	137

Chg in ALK (mg/L)

Run 1	7.3
Run 2	4
Run 3	3
Run 7	14

Nitrate Use (mg/L)

Run 1	2
Run 2	1.1
Run 3	0.8
Run 7	3.9

Runs with Acetate

	Sample size	Start pt	End pt	Titrant used	Total ALK
Run 4 Infl	200	3.7	44	40.3	201.5
Run 5 Infl	200	0.9	41.4	40.5	202.5
Run 6 Infl	200	6.5	46.9	40.4	202
Run 8 Infl	100	13.6	27.9	14.3	143
Run 9 Infl	100	27.9	43.1	15.2	152
Run 4 Effl	200	0.7	43.4	42.7	213.5
Run 5 Effl	200	2.1	44.6	42.5	212.5
Run 6 Effl	200	4.1	45.2	41.1	205.5
Run 8 Effl	100	1.7	18.9	17.2	172
Run 9 Effl	100	18.9	34.8	15.9	159

Chg in ALK (mg/L)

Run 4	12
Run 5	10
Run 6	3.5
Run 8	29
Run 9	7

Nitrate Use (mg/L)

Run 4	3.4
Run 5	2.8
Run 6	1
Run 8	8.1
Run 9	2

Location: Chamber's Creek
Date: **20-Aug-98**

Run 1 Effl	SCOD	49.8
	FCOD	28.6
	RBCOD	5.9
	TBSCOD	25.2
	BSCOD	19.3

Run 2 Effl	SCOD	78.2
	FCOD	33.3
	RBCOD	10.6
	TBSCOD	53.6
	BSCOD	43

Run 3 Effl	SCOD	80.1
	FCOD	38.8
	RBCOD	16.1
	TBSCOD	55.5
	BSCOD	39.4

WW (Runs 1-3	TCOD	216
	SCOD	66
	FCOD	51.8
	RBCOD	29.1
	TBSCOD	41.4
	BSCOD	12.3

MLSS Runs 1-3	SCOD	234
	TBSCOD	209.4

Run 4 Effl	SCOD	67.2
	FCOD	42
	RBCOD	19.3
	TBSCOD	42.6
	BSCOD	23.3

Run 5 Effl	SCOD	88.2
	FCOD	52.7
	RBCOD	30
	TBSCOD	63.6
	BSCOD	33.6

Run 6 Effl	SCOD	91.3
	FCOD	58.3
	RBCOD	35.6
	TBSCOD	66.7
	BSCOD	31.1

WW Runs (3-6	TCOD	279
	SCOD	131.7
	FCOD	99.7
	RBCOD	77
	TBSCOD	109
	BSCOD	32

MLSS Runs 4-	SCOD	236
	TBSCOD	211.4

Location: Chamber's Creek
Date: **24-Sep-98**

Run 7 Effl	SCOD	38.7
	FCOD	34.6
	RBCOD	11.9
	TBSCOD	14.1
	BSCOD	2.2

Run 7 Infl	SCOD	51.3
	FCOD	37.5
	RBCOD	14.8
	TBSCOD	26.7
	BSCOD	11.9

Run 9 Effl	SCOD	86.6
	FCOD	72.9
	RBCOD	50.2
	TBSCOD	62
	BSCOD	11.8

Run 9 Infl	SCOD	103.1
	FCOD	86.6
	RBCOD	63.9
	TBSCOD	78.5
	BSCOD	14.6

Run 8 Effl	SCOD	49.1
	FCOD	48.5
	RBCOD	25.8
	TBSCOD	24.5
	BSCOD	-1.3

Run 8 Infl	SCOD	90.3
	FCOD	71.2
	RBCOD	48.5
	TBSCOD	65.7
	BSCOD	17.2

PLANT EFFLUENT (20 Aug 98)

FCOD	22.7

PLANT EFFLUENT (24 Sep 98)

SCOD	24.6
FCOD	22.7

VSS/TSS ANALYSIS

Location: Chamber's Creek
Date: #######

	tare weigh	105 C	550 C	TSS(mg/L)	VSS(mg/L	Samp Sz	VSS/TSS
ML Runs 1-3	1.4235	1.478	1.4384	5470	3980	10	0.73
ML Runs 1-3	1.433	1.488	1.4478	5490	4010	10	0.73
ML Runs 4-6	1.4346	1.483	1.4441	4840	3890	10	0.8
ML Runs 4-6	1.4312	1.48	1.4408	4850	3890	10	0.8
WW Runs 1-3	1.427	1.428	1.4272	30	23.3	30	0.78
WW Runs 1-3	1.4256	1.426	1.4259	26.67	16.7	30	0.63
WW Runs 4-6	1.4276	1.429	1.428	36.67	23.3	30	0.64
WW Runs 4-6	1.4301	1.431	1.4304	20	10	30	0.5

ML Runs 1-3	
VSS (mg/L)	TSS (mg/L)
3995	5480

ML Runs 4-6	
VSS (mg/L	TSS (mg/L)
3890	4845

ML AVE
VSS/TSS
0.76

WW Runs 1-3	
VSS (mg/L)	TSS (mg/L)
20	28.3

WW Runs 4-6	
VSS (mg/L	TSS (mg/L)
16.7	28.3

WW AVE
VSS/TSS
0.65

Location: Chamber's Creek
Date: #######

	tare weigh	105 C	550 C	TSS(mg/L)	VSS(mg/L	Samp Sz
ML Run 7	1.4171	1.445	1.4206	1886.7	1653.3	15
ML Run 7	1.4198	1.449	1.4232	1920	1693.3	15
ML Run 8	1.4044	1.437	1.4098	2160	1800	15
ML Run 9	1.4064	1.438	1.4107	2113.3	1826.7	15

ML Run 7 Ave	
VSS (mg/L)	TSS (mg/L)
1673.3	1903.35

ML AVE
VSS/TSS
0.86

MLSS ANALYSIS

Location	Chamber's Creek
Date of sample	20-Aug-98

	tare weight	105 C	550 C	TSS	VSS	Samp Sz (mL)
Sample 1	1.4329	1.464	1.4373	6220	5340	5
Sample 2	1.4296	1.4603	1.4339	6140	5280	5

MLSS characteristics
After Aeration

	Test #1	Test #2	Average
TSS	6220	6140	6180
VSS	5340	5280	5310

2.86	$g\ O_2 / g\ NO_3$	
0.8	$g\ O_2 / g$ COD	
1.42	$g\ O_2 / g$ biomass	

Fraction which can Denitrify

Time (min)	Sample A DO Conc (mg/L)
0	9.05
1	7.63
2	6.43
3	5.25
4	4.09
5	2.93
6	1.76
7	0.63
8	0.48
9	0.45
10	0.43
15	0.37
20	0.33
25	0.3
30	0.3

Temperature:	17.8

Temp Correction Factor:
$OUR_{20}=OUR_T/(@^{T-20})$

@=	1.029

$SEOUR_{NC}$:	0.3184
$SEOUR_{TC}$:	0.339

Endogenous Decay Coefficient for O2 use (b):	0.24

Time (min)	Sample A NO_3 Conc (mg/L)
0	10.7
20	10.6
40	7.33
60	5
80	2.3
100	0.5
120	

Temperature:	21.6

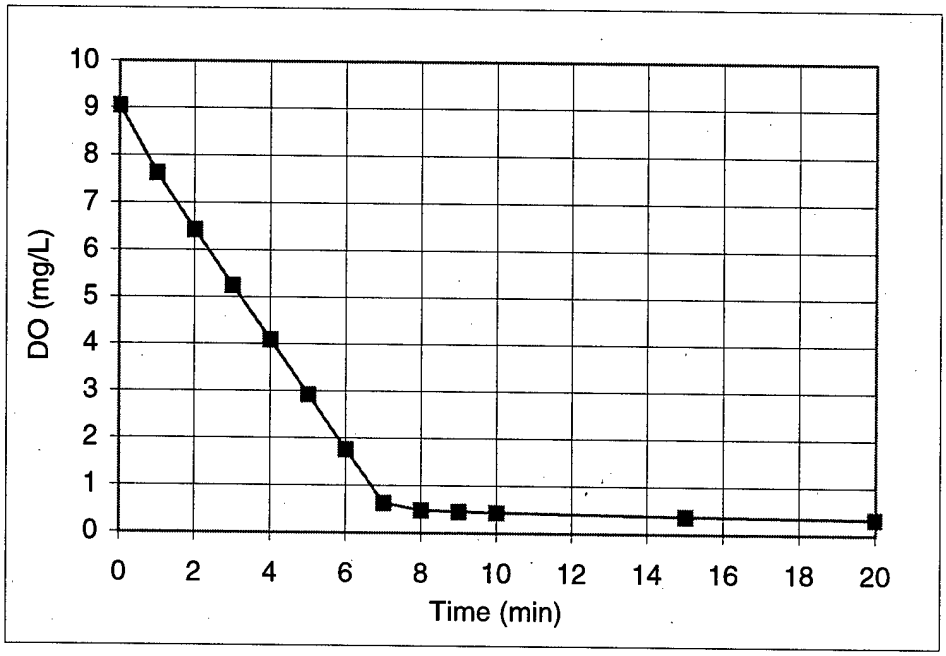
Temp Correction Factor:
$SDNR_{20}=(SDNR_T)/(@^{T-20})$

@=	1.029

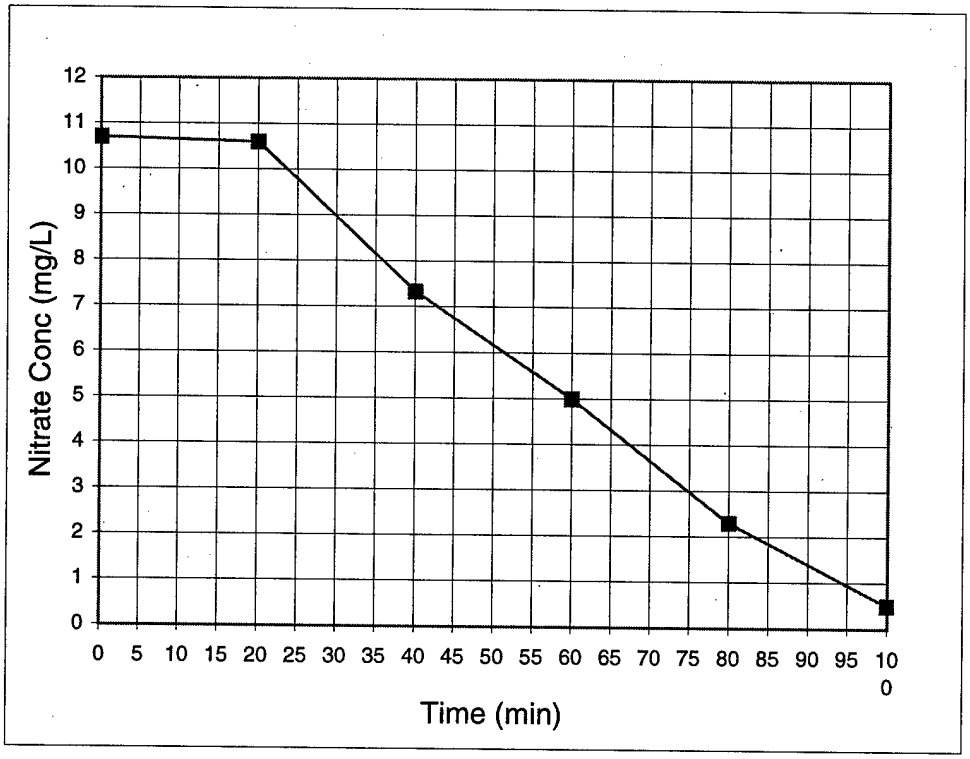
$SDNR_{NC}$:	0.0375
$SDNR_{TC}$:	0.0358

Fraction Denitrifiying:	0.302

Endogenous Decay Coefficient for NO3 use (b):	0.07



Slope: -1.17 Intercept 8.78



Slope: -0.14 Intercept: 13.12

Active Biomass Calaculations

Location: Chamber's Creek WWTP
Data Used: August Monthly Discharge Report

Reported Data

Y (g/g-d)	1.08		V (MG)	3.38		
MCRT (d)	3.6		Q (MGD)	13.61		
$CBOD_{FE}$ (mg/l	3.3		$CBOD_{PE}$ (mg/L)	106		
$CBOD_{FE}$ (lb/d)	373	374.6	$CBOD_{PE}$ (lb/d)	11830	12031	
AB (lbs/d)	44759		$CBOD_{PE}/CBOD$	0.46		
WAS (lbs/d)	12433.1		$CBOD_{FE}/CBOD$	0.13		

Formulas

X_T	Px*(SRT)/V
Px_{bio}	Y_{net}*(chg BOD)*Q
Ynet	Y/(1+b*SRT)
Y_{obs}	Px/(chg BOD)

b	0.08	
conversion	8.34	lb/MG-mg/L
Y	0.6	g TSS/g BOD

Calculations

Y_{net}	0.47
Y_{obs}	1.03
$X_{T\ (mg/L)}$	1587.8
$Px_{bio\ (lb/d)}$	5478.89
X_{bio}/X_T	0.44

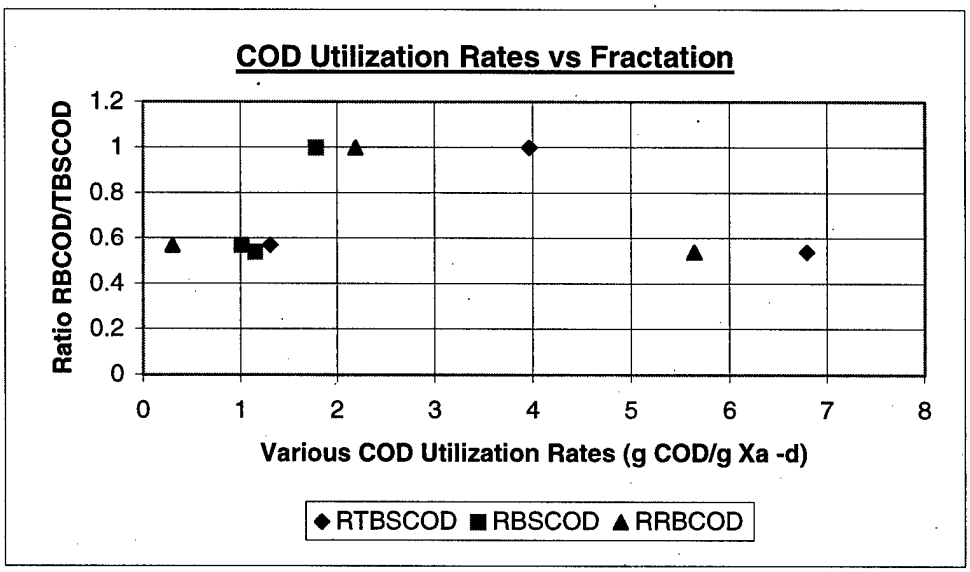
SDNR vs RBCOD
Chamber's Creek WWTP
Date: 20 Aug 98 & 24 Sep 98

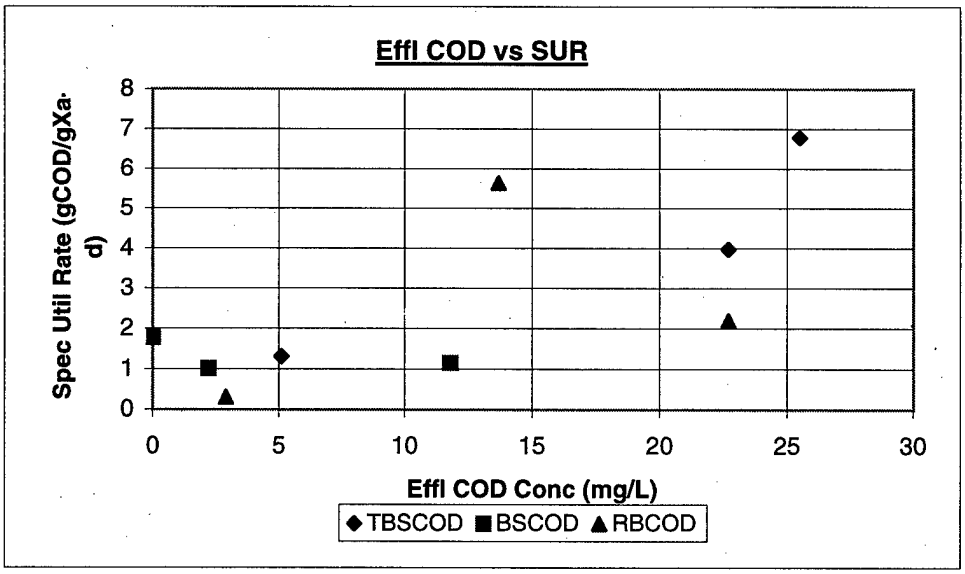
2.86	g O_2 / g NO_3
0.8	g O_2 / g COD
0.07	g $X_{d\ death}$/gX-d
1.42	g O_2 / g biomass

RUN	HRT	Temp	effl RBCOD	chg RBCOD	chg BSCOD	SDNR-NC	SDNR-TC	$SDNR_{AVSS}$	$SDNR_{ADLSS}$
	(min)	(°C)	(mg/L)	(mg/L)	(mg/L)	(gNO3/gVSS-d	(gNO3/gVSS-d)	(gNO3/gAVSS-d)	(gNO3/gADVSS-d)
Run 1	30.2	27	5.9	14.1	68.5	0.083	0.068	0.155	0.513
Run 2	20.7	26.8	10.6	8.7	44.5	0.066	0.055	0.125	0.414
Run 3	14.7	27	16.1	2.4	47.2	0.067	0.055	0.125	0.414
Run 4	29.5	25.2	19.3	31.4	101.3	0.128	0.11	0.25	0.828
Run 5	21.4	25	30	19.9	81.5	0.131	0.114	0.259	0.858
Run 6	14.1	24.6	35.6	15.7	76.5	0.101	0.089	0.202	0.669
Run 7	62.5	18	11.9	2.9	9.7	0.063	0.067	0.152	0.503
Run 8	62.5	18.8	25.8	22.7	18.5	0.112	0.116	0.264	0.874
Run 9	14.4	19	50.2	13.7	2.8	0.109	0.113	0.257	0.851

RUN	HRT	VSS	effl BSCOD	effl RBCOD	chg NO3	NO3 for respiration	NO3 for COD_{use}	R_{BSCOD}	R_{RBCOD}
	(min)	(mg/L)	(mg/L)	(mg/L)	(mg/L)	(mg/L)	(mg/L)	(gBSCOD/gXa-d)	(gRBCOD/gXa-d)
Run 1	30.2	1257	19.3	14.1	2.2	0.92	1.28	19.56	4.03
Run 2	20.7	1362	43	8.7	1.3	0.68	0.62	17.11	3.35
Run 3	14.7	1471	39.4	2.4	1	0.52	0.48	23.65	1.2
Run 4	29.5	1338	23.3	31.4	3.5	0.95	2.55	27.81	8.62
Run 5	21.4	1382	33.6	19.9	2.7	0.71	1.99	29.86	7.29
Run 6	14.1	1310	31.1	15.7	1.3	0.45	0.85	44.9	9.21
Run 7	62.5	1673	2.2	2.9	4.6	2.52	2.08	1.01	0.3
Run 8	62.5	1800	0	22.7	8.75	2.72	6.03	1.78	2.19
Run 9	14.4	1827	11.8	13.7	2	0.63	1.37	1.15	5.64

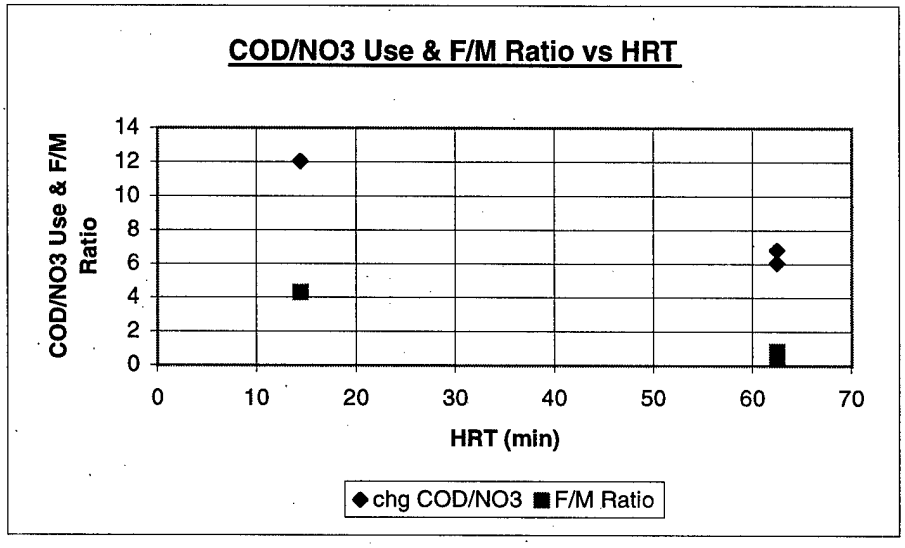
Run	effl TBSCOD (mg/L)	effl BSCOD (mg/L)	effl RBCOD (mg/L)	R_{TBSCOD} (gCOD/gXa-d)	R_{BSCOD} (gCOD/gXa-d)	R_{RBCOD} (gCOD/gXa-d)	$R_{RBCOD/TBSCOD}$ (g/g)
Run 1	33.4	19.3	14.1	23.59	19.56	4.03	0.42
Run 2	51.7	43	8.7	20.46	17.11	3.35	0.17
Run 3	41.8	39.4	2.4	24.85	23.65	1.2	0.06
Run 4	54.7	23.3	31.4	36.43	27.81	8.62	0.57
Run 5	53.5	33.6	19.9	37.15	29.86	7.29	0.37
Run 6	46.8	31.1	15.7	54.11	44.9	9.21	0.34
Run 7	5.1	2.2	2.9	1.31	1.01	0.3	0.57
Run 8	22.7	0	22.7	3.97	1.78	2.19	1
Run 9	25.5	11.8	13.7	6.79	1.15	5.64	0.54

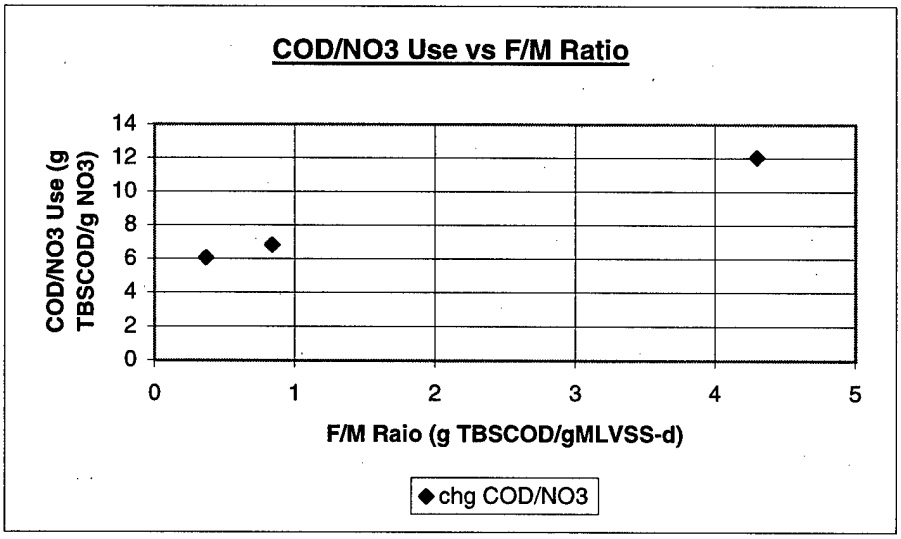




Chamber's Creek Experiments

	chg TBSCOD	NO3 for COD_{use}	chg COD/NO_3 (mg TBSCOD	F/M Ratio (g TBSCOD	HRT
	(mg TBSCOD/L	(mg/L)	/mg NO3)	/g MLVSS-d)	(min)
Run 7	12.6	2.08	6.06	0.37	62.5
Run 8	41.2	6.03	6.83	0.84	62.5
Run 9	16.5	1.37	12.04	4.3	14.4





Plant Specific General Operations Information

LOCATION: Chamber's Creek WWTP Steilacoom, WA

Parameter	*June Ave*	*July Ave*	*August Ave*	*Units*
MLSS	1546	1558	1587	mg/L
MLVSS	1237	1278	1301	mg/L
Daily Flow	13.75	13.51	13.61	MGD
MCRT	4.4	4.3	4.4	d^{-1}
F/M Ratio	0.39	0.37	0.32	lb BOD/lb MLVSS-d
Hetertrophic Yield	0.98	1.13	1.08	g/g
AB SVI	201	189	166	
WWSS	276	270	219	mg/L
WWVSS	245	242	194	mg/L
Raw Infl TBOD conc	228	274	191	mg/L
Raw Infl COD conc	528	551	533	mg/L
Raw Infl NH_3	26.9	27.1	31.7	mg/L
Raw Infl NO_3	n/a	n/a	n/a	mg/L
Raw Infl NO_2	n/a	n/a	n/a	mg/L
Raw Infl pH	7.2	7.2	7.2	
Raw Infl DO	1.1	1.2	0.6	mg/L
Raw Infl Temp	17.6	18.8	20.2	° C
Raw Infl Alk	170	200	230	mg/L (as $CaCO_3$)
Final Effl COD	33	38	41	mg/L
Final Effl NH_3	27.2	23.4	28.4	mg/L
Final Effl NO_3	0.02	0.01	0.11	mg/L
Final Effl NO_2	0.58	0.66	0.66	mg/L
Final Effl PO_4	n/a	n/a	n/a	mg/L
Final Effl ALK	170	188	244	mg/L (as $CaCO_3$)
Final Effl CBOD	8.3, 946	6.7, 743	3.3, 373	mg/L, lbs/d
Prim Effl CBOD	109, 12431	114, 12714	106, 11830	mg/L, lbs/d
AB LB	42017	41729	44759	lbs/d
Sludge Production	12215	12739	12614	lbs/d

Analytical Report (LOTT WWTP) 25,27 August 1998

Run 1
no acetate

WW	Average
TSS	193.3
VSS	146.7
TCOD	688
TBSCOD	304.8
RBCOD	28.5

Temp	20.8
pH	7.27
ORP	-33
HRT	33.3
Q-WW	45.7
Q-MLSS	14.3
Qtot	60

MLSS	Average
TSS	5390
VSS	4250
TBSCOD	60.5

WW1

Temp	20.2
pH	6.74
ORP	-3.8

Influent	Average
Ortho-P	31.2
Alkalinity	197
NO3-N-N	14.3

Calculated:

Infl-TSS	1432
Infl-VSS	1125
Infl-TBSCOD	246.6
Infl-RBCOD	21.7
chg NO3-N	7.1
SDNR-TC	0.267
chg TBSCOD	199.3
chg RBCOD	3.7
chg ALK	24.5
SDNR-NC	0.273

MLSS1

Temp	21
pH	6.46
ORP	13.9

Effluent	Average
TBSCOD	47.3
RBCOD	18
NO3-N-N	7.2
Ortho-P	34
Alkalinity	221.5

Run 2
no acetate

Influent	Average
Ortho-P	
Alkalinity	200
NO3-N-N	11.6

Calculated:

Infl-TSS	1945
Infl-VSS	1530
Infl-TBSCOD	222.5
Infl-RBCOD	18.9
chg NO3-N	6.8
SDNR-TC	0.277
chg TBSCOD	165.5
chg RBCOD	-7.6
chg ALK	20
SDNR-NC	0.284

Temp	20.9
pH	7.21
ORP	-29
HRT	22.5
Q-WW	59
Q-MLSS	30
Qtot	89

Effluent	Average
TBSCOD	57
RBCOD	26.5
NO3-N-N	4.8
Ortho-P	
Alkalinity	220

WW1

Temp	20.5
pH	6.79
ORP	-6.7

MLSS1	Average
Temp	23.5
pH	6.95
ORP	-18.6

Run 3
no acetate

Temp	21.1
pH	7.13
ORP	-24
HRT	15.7
Q-WW	93.5
Q-MLSS	33.9
Qtot	127

WW1

Temp	20.9
pH	6.84
ORP	-9.2

Influent	Average
Ortho-P	25.9
Alkalinity	217
NO3-N-N	13.5

Effluent	Average
TBSCOD	64.9
RBCOD	45
NO3-N-N	9.5
Ortho-P	
Alkalinity	231

MLSS1	Average
Temp	21.4
pH	7.16
ORP	-26.6

Calculated:

Infl-TSS	1576
Infl-VSS	1239
Infl-TBSCOI	239.8
Infl-RBCOD	20.9
chg NO3-N	4
SDNR-TC	0.287
chg TBSCOI	174.9
chg RBCOD	-24.1
chg ALK	14
SDNR-NC	0.296

Run 4
with acetate

Temp	20
pH	7.29
ORP	-37
HRT	30.3
Q-WW	46.4
Q-MLSS	19.6
Qtot	66

WW1

Temp	20.8
pH	6.81
ORP	-9.4

MLSS1

Temp	20.1
pH	7.16
ORP	-29

WW	Average
TSS	145
VSS	128.4
TCOD	828
TBSCOD	550.8
RBCOD	76.4

MLSS	Average
TSS	7320
VSS	5760
TBSCOD	77.6

Influent	Average
Ortho-P	36.2
Alkalinity	282
NO3-N-N	22

Effluent	Average
TBSCOD	129
RBCOD	105.9
NO3-N-N	9.2
Ortho-P	49.1
Alkalinity	326.7
Nitrite	2.435

Calculated:

Infl-TSS	2276
Infl-VSS	1801
Infl-TBSCOI	410.3
Infl-RBCOD	53.7
chg NO3-N	12.8
SDNR-TC	0.338
chg TBSCOI	281.3
chg RBCOD	-52.2
chg ALK	44.7
SDNR-NC	0.338

Run 5
with acetate

Temp	21.8
pH	7.26
ORP	-35
HRT	21.9
Q-WW	62.6
Q-MLSS	28.6
Qtot	91.2

Influent	Average
Ortho-P	32
Alkalinity	285.3
NO3-N-N	20.8

Effluent	Average
TBSCOD	140.3
RBCOD	110
NO3-N-N	10.1
Ortho-P	42.8
Alkalinity	321.3

Calculated:

Infl-TSS	2395
Infl-VSS	1895
Infl-TBSCOI	402.4
Infl-RBCOD	52.4
chg NO3-N	10.7
SDNR-TC	0.353
chg TBSCOI	262.1
chg RBCOD	-57.6
chg ALK	36
SDNR-NC	0.371

WW1

Temp	20.8
pH	6.83
ORP	-11

MLSS1

Temp	20.2
pH	7.24
ORP	-33.4

Run 6
with acetate

Temp	22.3
pH	7.2
ORP	-31
HRT	16.2
Q-WW	88.2
Q-MLSS	35.3
Qtot	124

Influent	Average
Ortho-P	25.9
Alkalinity	297.3
NO3-N-N	20.2
Nitrite	2.255

Effluent	Average
TBSCOD	152
RBCOD	118.2
NO3-N-N	11.9
Ortho-P	34.8
Alkalinity	328.7

Calculated:

Infl-TSS	2196
Infl-VSS	1738
Infl-TBSCOI	415.5
Infl-RBCOD	54.6
chg NO3-N	8.3
SDNR-TC	0.397
chg TBSCOI	263.5
chg RBCOD	-63.6
chg ALK	31.4
SDNR-NC	0.424

WW1

Temp	21
pH	6.86
ORP	-12

MLSS1

Temp	20.5
pH	7.3
ORP	-37.2

SNDR Corr Factor:	1.029

Analytical Report (LOTT) 22 September 1998

Run 7
no acetate

Temp	19.1
pH	7.85
ORP	-54.4
HRT	59.3
Q-WW	24.5
Q-MLSS	9.2
Qtot	33.7
DO	0.3

WW1

Temp	19.2
pH	7.4
ORP	-28.3
DO	0.6

MLSS1

Temp	18.9
pH	7
ORP	-9.4
DO	0.7

Influent	Average	
TBSCOD	49.9	
RBCOD	36	
Ortho-P	29.1	
Alkalinity	206	
NO3-N	23	23.3
Nitrite	2.22	

Effluent	Average	
TBSCOD	9.8	
RBCOD	11	
NO3-N	14.3	14.2
Nitrite	3.53	
Ortho-P	24.1	
Alkalinity	237	

Calculated:

Infl-TSS	1487
Infl-VSS	1227
chg BSCOD	40.1
chg RBCOD	25
chg ALK	31
chg NO3-N	8.9
SDNR-TC	0.181
SDNR-NC	0.176

Run 8
no acetate

Temp	20.2
pH	7.53
ORP	-37.7
HRT	14.7
Q-WW	96.7
Q-MLSS	39.3
Qtot	136

WW1

Temp	20.1
pH	7.37
ORP	-29.6

Influent	Average	
TBSCOD	45.5	
RBCOD	30.3	
Ortho-P	29.1	
Alkalinity	214	
NO3-N	22.8	23.1
Nitrite		

Effluent	Average	
TBSCOD	21	
RBCOD	16.7	
NO3-N	18.8	19
Nitrite		
Ortho-P	23.2	
Alkalinity	230	

Calculated:

Infl-TSS	1573
Infl-VSS	1280
chg TBSCOD	24.5
chg RBCOD	13.6
chg ALK	16
chg NO3-N	4.05
SDNR-TC	0.308
SDNR-NC	0.31

Analytical Report (LOTT) 22 September 1998

Run 9
with acetate

Temp	21.1
pH	8.08
ORP	-66.3
HRT	63.9
Q-WW	21.6
Q-MLSS	9.7
Qtot	31.3

WW1

Temp	21
pH	7.26
ORP	-22.4

MLSS1

Temp	20.9
pH	7.22
ORP	-20.4

Influent	Average	
TBSCOD	76.9	
RBCOD	55.3	
Ortho-P	32.9	
Alkalinity	268	
NO3-N	24.4	25
Nitrite	2.51	

Effluent	Average	
TBSCOD	25.5	
RBCOD	33.5	
NO3-N	3.6	3.5
Nitrite	2.93	
Ortho-P	45.8	
Alkalinity	342	

Calculated:

Infl-TSS	1793
Infl-VSS	1453
chg TBSCOD	51.4
chg RBCOD	21.8
chg ALK	74
chg NO3-N	21.15
SDNR-TC	0.318
SDNR-NC	0.328

Run 10
with acetate

Temp	22
pH	7.53
ORP	-37.3
HRT	14
Q-WW	101.8
Q-MLSS	41.1
Qtot	142.9

WW1

Temp	21.2
pH	7.26
ORP	-23.7

Influent	Average	
TBSCOD	89.4	
RBCOD	75.2	
Ortho-P	26.2	
Alkalinity	276	
NO3-N	25.3	24.9
Nitrite		

Effluent	Average	
TBSCOD	58.9	
RBCOD	49.3	
NO3-N	19.3	18.8
Nitrite		
Alkalinity	298	
Ortho-P	29.1	

MLSS1

Temp	21.3
pH	7.23
ORP	-23.4

Calculated:

Infl-TSS	1900
Infl-VSS	1580
chg TBSCOD	30.5
chg RBCOD	25.9
chg ALK	22
chg NO3-N	6.05
SDNR-TC	0.372
SDNR-NC	0.394

SNDR Corr Factor:	1.029

PLANT EFFLUENT

Ortho-	10	
SCOD	31	
FCOD	23	23.1

Location: LOTT WWTP
Date: 25,27 Aug 98 & 22 Sep 98

Alkalinity(mg $CaCO_3$/L) to pH 4.3

Titrant	H_2SO_4
Normality	0.02

Runs w/o acetate

	Sample size	Start pt	End pt	Titrant used	Total ALK
Run 1 Infl	200	5.1	44.5	39.4	197
Run 2 Infl	200	3.2	43.2	40	200
Run 3 Infl	200	5.7	49.1	43.4	217
Run 7 Infl	100	3.3	23.9	20.6	206
Run 8 Infl	100	3	24.4	21.4	214
Run 1 Effl	200	3.5	47.8	44.3	221.5
Run 2 Effl	200	0.4	44.4	44	220
Run 3 Effl	200	2.7	48.9	46.2	231
Run 7 Effl	100	3.5	27.2	23.7	237
Run 8 Effl	100	3.2	26.2	23	230

Chg in ALK (mg/L)

Run 1	24.5
Run 2	20
Run 3	14
Run 7	31
Run 8	16

Nitrate Use (mg/L)

Run 1	6.9
Run 2	5.6
Run 3	3.9
Run 7	8.7
Run 8	4.5

Runs with Acetate

	Sample size	Start pt	End pt	Titrant used	Total ALK
Run 4 Infl	150	2.1	44.4	42.3	282
Run 5 Infl	150	2.6	45.4	42.8	285.3
Run 6 Infl	150	3.5	48.1	44.6	297.3
Run 9 Infl	100	0	26.8	26.8	268
Run 10 Infl	100	4.7	32.3	27.6	276
Run 4 Effl	150	0.4	49.4	49	326.7
Run 5 Effl	150	1.2	49.4	48.2	321.3
Run 6 Effl	150	0.1	49.4	49.3	328.7
Run 9 Effl	100	2.6	36.8	34.2	342
Run 10 Effl	100	3.6	33.4	29.8	298

Chg in ALK (mg/L)

Run 4	44.7
Run 5	36
Run 6	31.4
Run 9	74
Run 10	22

Nitrate Use (mg/L)

Run 4	12.5
Run 5	10.1
Run 6	8.8
Run 9	20.7
Run 10	6.2

Location: LOTT WWTP
Date: **25,27 Aug 98**

COD ANALYSIS

Run 1 Effl	SCOD	78.5
	FCOD	41.3
	RBCOD	18
	TBSCOD	47.3
	BSCOD	29.3

Run 2 Effl	SCOD	88.2
	FCOD	49.8
	RBCOD	26.5
	TBSCOD	57
	BSCOD	30.5

Run 3 Effl	SCOD	96.1
	FCOD	68.3
	RBCOD	45
	TBSCOD	64.9
	BSCOD	19.9

WW (Runs 1-3	TCOD	688
	SCOD	336
	FCOD	51.8
	RBCOD	28.5
	TBSCOD	304.8
	BSCOD	276.3

MLSS Runs 1-	SCOD	83.8
	TBSCOD	60.5

Run 4 Effl	SCOD	160.2
	FCOD	129.2
	RBCOD	105.9
	TBSCOD	129
	BSCOD	23.1

Run 5 Effl	SCOD	171.5
	FCOD	133.3
	RBCOD	110
	TBSCOD	140.3
	BSCOD	30.3

Run 6 Effl	SCOD	183.2
	FCOD	141.5
	RBCOD	118.2
	TBSCOD	152
	BSCOD	33.8

WW Runs (3-	TCOD	828
	SCOD	582
	FCOD	99.7
	RBCOD	76.4
	TBSCOD	550.8
	BSCOD	474.4

MLSS Runs 1	SCOD	77.6
	TBSCOD	54.5

Location: LOTT WWTP
Date: **22-Sep-98**

Run 7 Effl	SCOD	41
	FCOD	34.1
	RBCOD	11
	TBSCOD	9.8
	BSCOD	-1.2

Run 7 Infl	SCOD	81.1
	FCOD	59.1
	RBCOD	36
	TBSCOD	49.9
	BSCOD	13.9

Run 8 Effl	SCOD	52.2
	FCOD	39.8
	RBCOD	16.7
	TBSCOD	21
	BSCOD	4.3

Run 9 Effl	SCOD	56.7
	FCOD	56.6
	RBCOD	33.5
	TBSCOD	25.5
	BSCOD	-8

Run 9 Infl	SCOD	108.1
	FCOD	78.4
	RBCOD	55.3
	TBSCOD	76.9
	BSCOD	21.6

Run 10 Effl	SCOD	90.1
	FCOD	72.4
	RBCOD	49.3
	TBSCOD	58.9
	BSCOD	9.6

COD ANALYSIS

Run 8 Infl	SCOD	76.7
	FCOD	53.4
	RBCOD	30.3
	TBSCOD	45.5
	BSCOD	15.2

Run 10 Infl	SCOD	120.6
	FCOD	98.3
	RBCOD	75.2
	TBSCOD	89.4
	BSCOD	14.2

PLANT EFFLUENT (25 Aug 98)

FCOD	23.3

PLANT EFFLUENT (22 Sep 98)

SCOD	31.2
FCOD	23.1

VSS/TSS ANALYSIS

Location: LOTT WWTP
Date: **25,27 Aug 98**

	tare weight	105 C	550 C	TSS(mg/L)	VSS(mg/L)	Samp Sz	VSS/TSS
ML Runs 1-3	1.4271	1.45	1.4328	5400	4260	5	0.79
ML Runs 1-3	1.429	1.46	1.4347	5380	4240	5	0.79
ML Runs 4-6	1.4395	1.48	1.4473	7280	5720	5	0.79
ML Runs 4-6	1.4503	1.49	1.4581	7360	5800	5	0.79
WW Runs 1-3	1.4355	1.44	1.4369	193.33	146.7	30	0.76
WW Runs 1-3	1.4161	1.42	1.4175	193.33	146.7	30	0.76
WW Runs 4-6	1.4454	1.45	1.4459	146.67	130	30	0.89
WW Runs 4-6	1.4155	1.42	1.416	143.33	126.7	30	0.88

ML Runs 1-3	
VSS (mg/L)	TSS (mg/L)
4250	5390

ML Runs 4-6	
VSS (mg/L	TSS (mg/L)
5760	7320

ML AVE
VSS/TSS
0.79

WW Runs 1-3	
VSS (mg/L)	TSS (mg/L)
146.7	193.3

WW Runs 4-6	
VSS (mg/L	TSS (mg/L)
128.4	145

WW AVE
VSS/TSS
0.81

Location: LOTT WWTP
Date: **22-Sep-98**

	tare weight	105 C	550 C	TSS(mg/L)	VSS(mg/L)	Samp Sz	VSS/TSS
ML Run 7	1.4181	1.44	1.422	1486.7	1226.7	15	0.83
ML Run 8	1.4489	1.47	1.4533	1573.3	1280	15	0.81
ML Run 9	1.441	1.47	1.4461	1793.33	1453.3	15	0.81
ML Run 10	1.4389	1.47	1.4437	1900	1580	15	0.83

ML AVE
VSS/TSS
0.82

MLSS ANALYSIS

Location	LOTT WWTP
Date of sample	######

	tare wt	105 C	550 C	TSS	VSS	amp Sz (mL)
Sample 1	1.4506	1.4846	1.4635	6800	4220	5
Sample 2	1.4344	1.4686	1.4474	6840	4240	5

MLSS characteristics
After Aeration

	Test #1	Test #2	Average
TSS	6800	6840	6820
VSS	4220	4240	4230

2.86	g O_2 / g NO_3
0.8	g O_2 / g COD
1.42	g O_2 / g biomass

Fraction which can Denitrify

	Sample A
Time	DO Conc
(min)	(mg/L)
0	8.72
1	8.61
2	8.17
3	7.76
4	7.37
5	6.96
6	6.56
7	6.16
8	5.76
9	5.34
10	4.92
15	2.79
20	0.56
25	0.3
30	

Temperature:	17.7

Temp Correction Factor:
$OUR_{20}=OUR_T/(@^{T-20})$

@=	1.029

$SEOUR_N$	0.1396
$SEOUR_T$	0.1491

Endogenous Decay Coefficient for O2 use (b):	0.11

	Sample A
Time	NO_3 Conc
(min)	(mg/L)
0	13
25	13
50	9.6
75	6
100	1.8
125	0.9
150	

Temperature:	24.7

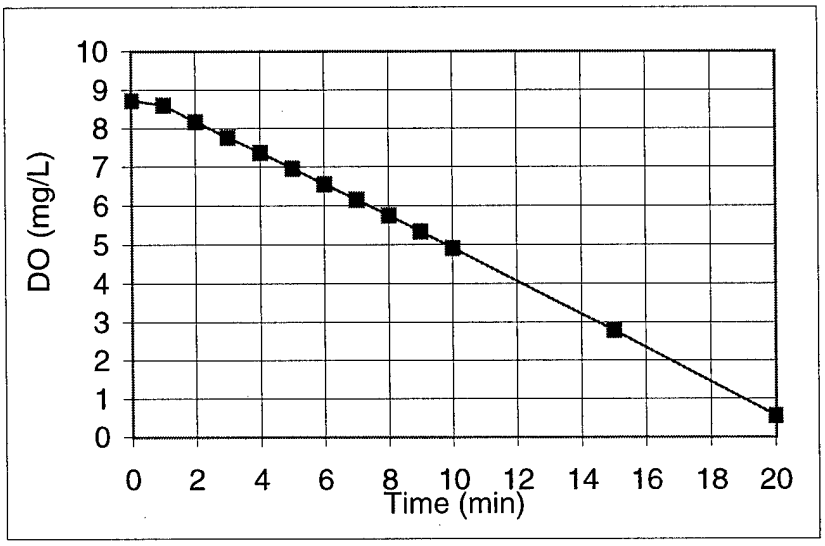
Temp Correction Factor:
$SDNR_{20}=(SDNR_T)/(@^{T-20})$

@=	1.029

$SDNR_N$	0.0508
$SDNR_T$	0.0444

Fraction Denitrifiyir	0.852

Endogenous Decay Coefficient for NO3 use (b):	0.09



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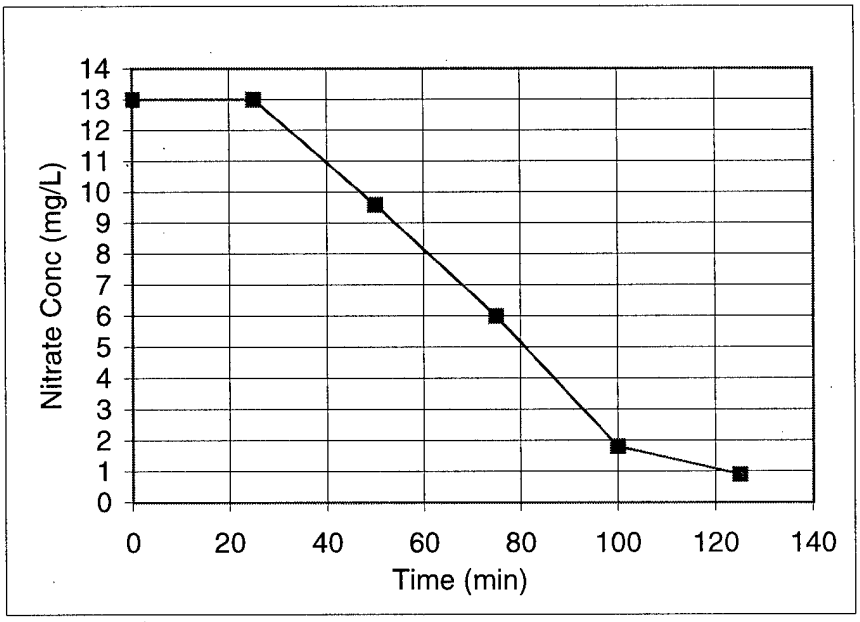

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Active Biomass Calculations

Location: LOTT WWTP
Data Used: August Monthly Discharge Report

Reported Data

Y (g/g-d)	0.681
MCRT (d)	11.6
$CBOD_{FE}$ (m	3.33
$CBOD_{FE}$ (lb	250
AB (lbs/d)	1E+05
WAS (lbs/d)	10495

V (MG)	6.45
Q (MGD)	8.85
$CBOD_{PE}$ (mg/l	214
$CBOD_{PE}$ (lb/d)	15919
$CBOD_{PE}$/CBO	0.46
$CBOD_{FE}$/CBO	0.13

Formulas

X_T	Px*(SRT)/V
X_{bio}	Y_{net}*(chg BOD)*Q
Ynet	Y/(1+b*SRT)
Y_{obs}	Px/(chg BOD)

b	0.1	
conversion	8.3	lb/MG-mg/L
Y	0.6	g TSS/g BOD

Calculations

Y_{net}	0.31
Y_{obs}	0.67
X_T	2263
X_{bio}	4820
X_{bio}/X_T	0.46

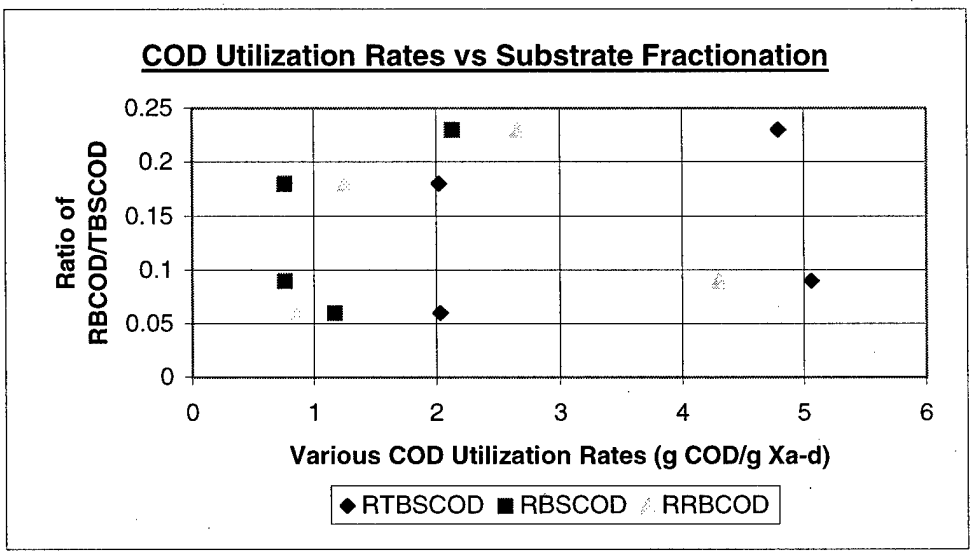
SDNR vs RBCOD
LOTT WWTP
Date: 25,27 Aug 98 & 22 Sep 98

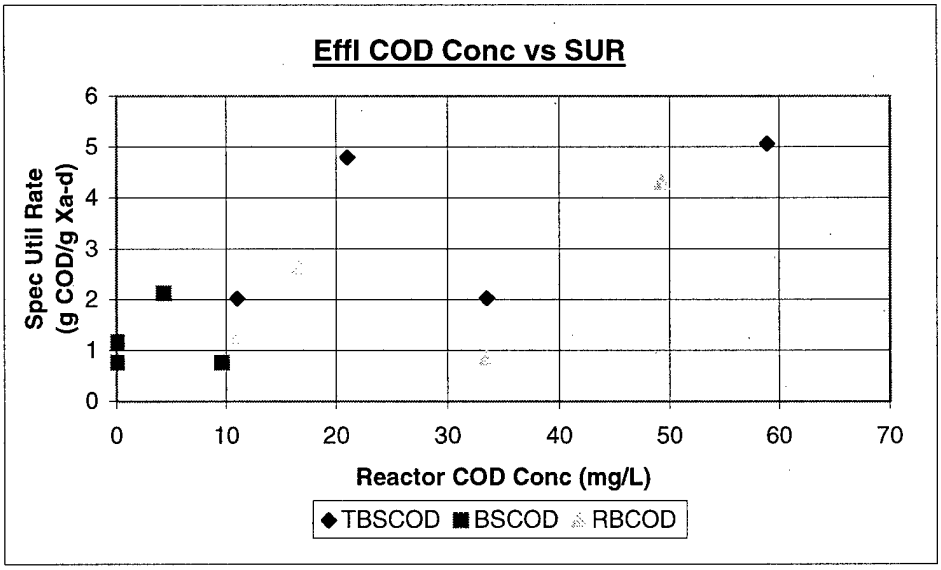
2.86	g O_2 / g NO_3
0.8	g O_2 / g COD
0.09	g X_d/gX-d
1.42	g O_2 / g biomass

RUN	HRT	Temp	effl RBCOD	chg RBCOD	chg BSCOD	SDNR-NC	SDNR-TC	$SDNR_{AVLSS}$	$SDNR_{FRAVLSS}$
	(min)	(°C)	(mg/L)	(mg/L)	(mg/L)	(gNO3/gVSS-d)	(gNO3/gVSS-d)	(gNO3/gAVSS-d)	(gNO3/gAVSS-d)
Run 1	33.3	20.8	18	3.7	199.3	0.273	0.267	0.58	0.681
Run 2	22.5	20.9	26.5	0	165.5	0.284	0.277	0.602	0.707
Run 3	15.7	21.1	45	0	174.9	0.296	0.287	0.624	0.732
Run 4	30.3	20	105.9	0	281.3	0.338	0.338	0.735	0.863
Run 5	21.9	21.8	110	0	262.1	0.371	0.353	0.767	0.9
Run 6	16.2	22.3	118.2	0	263.5	0.424	0.397	0.863	1.013
Run 7	59.3	19.1	11	25	15.1	0.176	0.181	0.393	0.461
Run 8	14.7	20.2	16.7	13.6	10.9	0.31	0.308	0.67	0.786
Run 9	63.9	21.1	33.5	21.8	29.6	0.328	0.318	0.691	0.811
Run 10	14	22	49.3	25.9	4.6	0.394	0.372	0.809	0.95

RUN	HRT	VSS	effl BSCOD	effl RBCOD	chg NO3	NO3 for respiration	NO3 for COD_{use}	R_{BSCOD}	R_{RBCOD}
	(min)	(mg/L)	(mg/L)	(mg/L)	(mg/L)	(mg/L)	(mg/L)	(gBSCOD/gXa-d)	(gRBCOD/gXa-d)
Run 1	33.3	1124.7	29.3	18	7.1	1.16	5.94	19.55	0.36
Run 2	22.5	1529.8	30.5	26.5	6.8	1.07	5.73	17.67	0
Run 3	15.7	1238.6	19.9	45	4	0.6	3.4	33.05	0
Run 4	30.3	1800.8	23.1	105.9	12.8	1.69	11.11	18.94	0
Run 5	21.9	1894.5	30.3	110	10.7	1.29	9.41	23.21	0
Run 6	16.2	1738.1	33.8	118.2	8.3	0.87	7.43	34.38	0
Run 7	59.3	1226.7	0	11	8.9	2.26	6.64	0.76	1.26
Run 8	14.7	1280	4.3	16.7	4.05	0.58	3.47	2.13	2.66
Run 9	63.9	1453.3	0	33.5	21.15	2.88	18.27	1.17	0.86
Run 10	14	1580	9.6	49.3	6.05	0.69	5.36	0.76	4.3

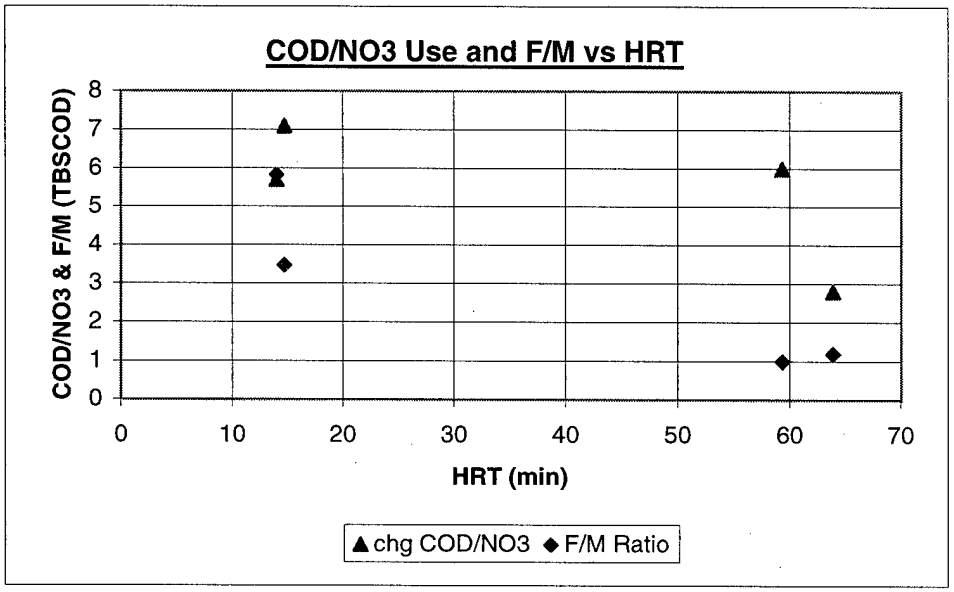
Run	effl TBSCOD (mg/L)	effl BSCOD (mg/L)	effl RBCOD (mg/L)	R_{TBSCOD} (gCOD/gXa-d)	R_{BSCOD} (gCOD/gXa-d)	R_{RBCOD} (gCOD/gXa-d)	$R_{RBCOD/TBSCOD}$ (g/g)
Run 1	47.3	29.3	18	19.91	19.55	0.36	0.42
Run 2	57	30.5	26.5	17.67	17.67	0	0.31
Run 3	64.9	19.9	45	33.05	33.05	0	0.51
Run 4	129	23.1	105.9	18.94	18.94	0	0.15
Run 5	140.3	30.3	110	23.21	23.21	0	0.17
Run 6	152	33.8	118.2	34.38	34.38	0	0.23
Run 7	11	0	11	2.02	0.76	1.26	0.18
Run 8	21	4.3	16.7	4.79	2.13	2.66	0.23
Run 9	33.5	0	33.5	2.03	1.17	0.86	0.06
Run 10	58.9	9.6	49.3	5.06	0.76	4.3	0.09

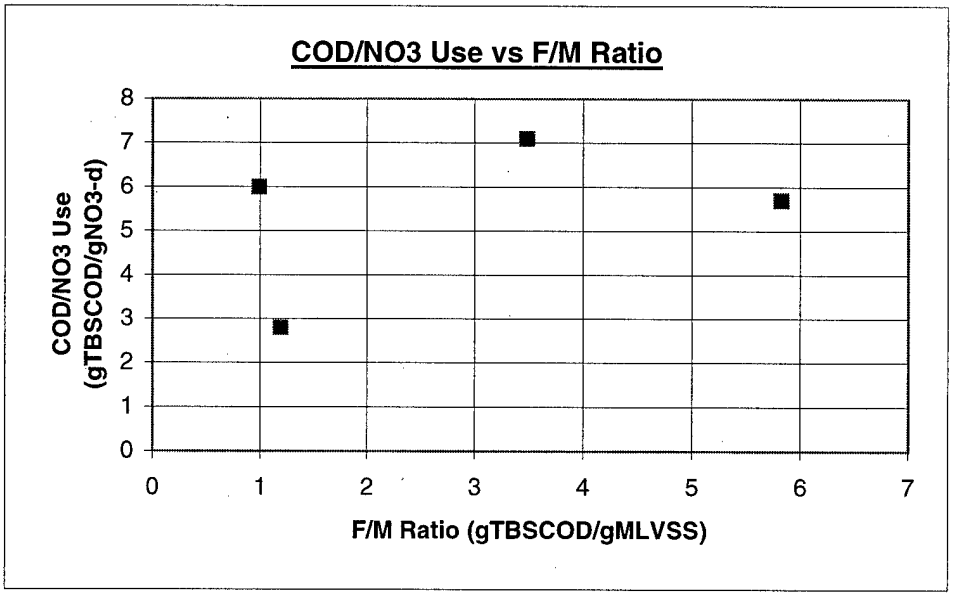




LOTT Experiments

	chg TBSCOD	NO3 for COD_{use}	chg COD/NO_3 (mg TBSCOD	F/M Ratio (g TBSCOD	HRT
	(mg TBSCOD/L)	(mg/L)	/mg NO3)	/g MLVSS-d)	(min)
Run 7	40.1	6.64	6	0.99	59.3
Run 8	24.5	3.47	7.1	3.48	14.7
Run 9	51.4	18.27	2.8	1.19	63.9
Run 10	30.5	5.36	5.7	5.82	14





Plant Specific General Operations Information

Location: LOTT WWTP Olympia, WA

Parameter	*Jun Ave*	*July Ave*	*August Ave*	*Units*
MLSS	1993	2271	2339	mg/L
MLVSS	1593	1714	1873	mg/L
Daily Flow	10.18	9.37	8.85	MGD
Aerobic SRT	10.8	11.2	11.6	d^{-1}
F/M Ratio (COD)	0.42	0.41	0.34	
Hetertrophic Yield	n.a.	0.654	0.681	g/g
MLSS SVI	75	63	85	
WWSS	242	255	269	mg/L
WWVSS	207	211	236	mg/L
Raw Infl TBOD cor	311	318	320	mg/L
Raw Infl COD conc	698	669	698	mg/L
Raw Infl NH_3	19	19.5	19.3	mg/L
Raw Infl NO_3	0.54	0.461	0.846	mg/L
Raw Infl NO_2	0.321	0.242	0.295	mg/L
Raw Infl pH	7	7	7	
Raw Infl DO	1.7	1.2	1.7	mg/L
Raw Infl Temp	17.3	19.1	19.1	$^{\circ}C$
Raw Infl Alk	191	201	191	mg/L (as $CaCO_3$)
Final Effl COD	37(94.5)	33(95)	32(96.2)	mg/L(% removal)
Final Effl NH_3	0.093	0.126	0.093	mg/L
Final Effl NO_3	1.73	1.54	1.52	mg/L
Final Effl NO_2	0.023	0.012	0.093	mg/L
Final Effl PO_4	2.84	3.08	n/a	mg/L
Final Effl ALK	133	144	139	mg/L (as $CaCO_3$)
Final Effl CBOD			2.72, 204	mg/L, lbs/d
Prim Effl CBOD			167, 12538	mg/L, lbs/d
AB LB			126312	lbs/d
Sludge Production			10495	lbs/d

Appendix 5 Description of Test Site Waste Water Treatment Plants

A. Olympus Terrace WWTP

The Olympus Terrace WWTP (OT WWTP) is located approximately 30 minutes North of Seattle in Muckleteo, Washington. The plant is located adjacent to the Puget Sound at the bottom of a wide ravine below the city of Muckleteo. The plant's NPDES permit (effective 1 Nov 98) allows the plant to discharge treated wastes into the Puget Sound meeting the effluent limitations in Table 4.1:

TABLE A5.1 Olympus Terrace Effluent Permit Requirements

Parameter	*Weekly Average*	*Monthly Average*
Flow		2.2 MGD
CBOD	40 mg/L, 759 lbs/day 85% Influent BOD removal	25 mg/L, 474 lbs/day 85% Influent BOD removal
TSS	45 mg/L, 854 lbs/day 85% Influent BOD removal	30 mg/L, 540 lbs/day 85% Influent BOD removal
Fecal Coliforms	400/100 mL	200/100 mL
PH	$9.0 \geq$ pH ≥ 6.0	
Total Residual Chlorine	325 µg/L	124 µg/L

Excursions between pH 5.0 and 6.0 or 9.0 and 10.0 are not violations as long as no single excursion exceeds 60 minutes in length and total excursions do not exceed 7 hours and 30 minutes per month. Any excursions below 5.0 and above 10.0 are violations. OT WWTP must also conduct acute toxicity tests and chronic toxicity tests twice during the five-year permit period. The acute toxicity tests consist of the Fathead minnow test (96 hour static-renewal test, method: EPA/600/4-90/027F) and the Daphnid, *Ceriodaphnia dubia, Daphnia pulex, Daphnia magna* test (48 hour static test, method: EPA/600/4-90/027F).

A.1 Treatment Processes used at Olympus Terrace

The overall plant treatment processes are summarized in Figure A5.1.

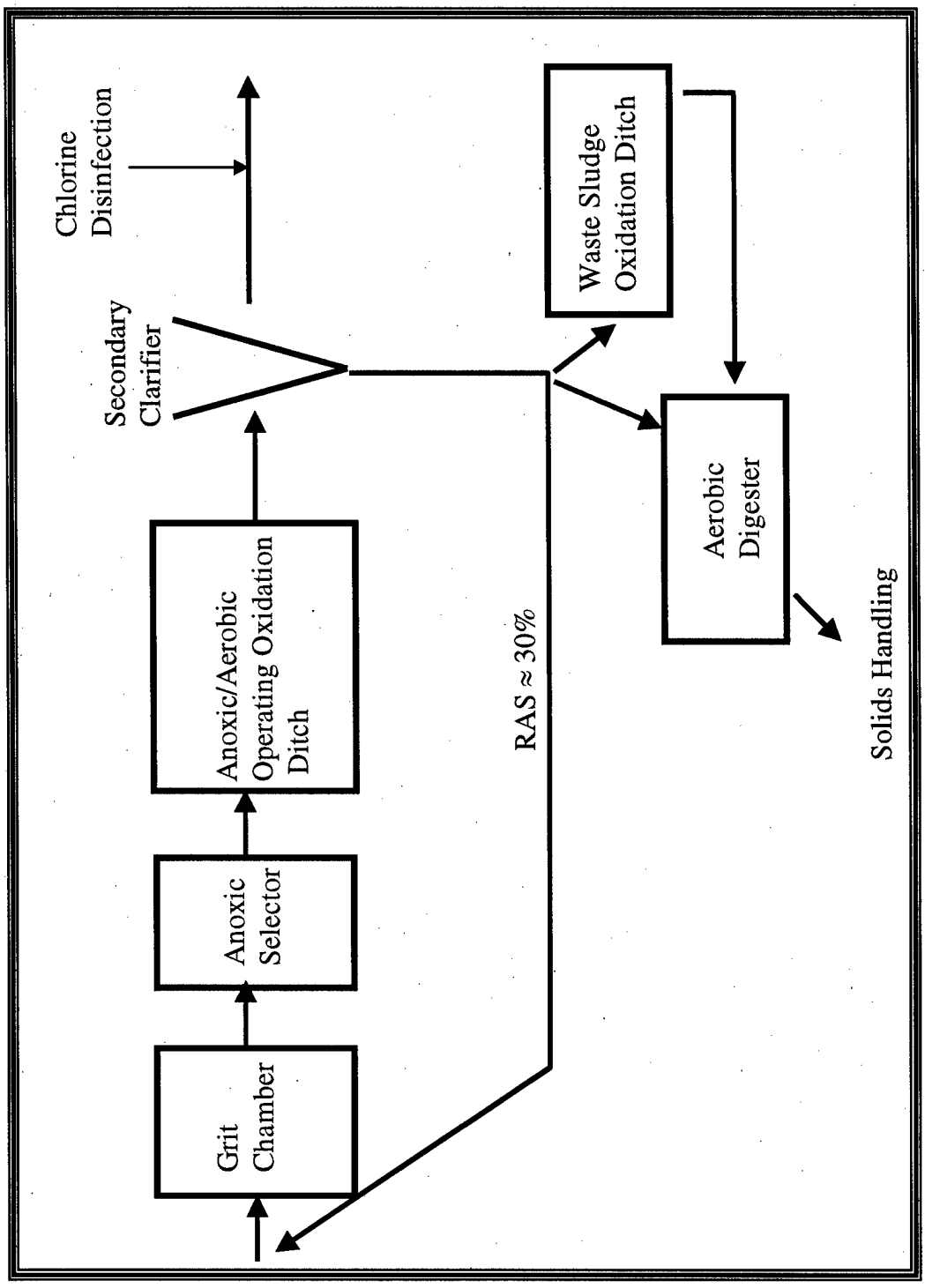


Figure 5A.1 Olympus Terrace WWTP Schematic

The activated sludge system is an oxidation ditch with two aerators. No primary treatment is used. Wastewater entering the plant is screened by one-inch bar screen before mixing with the return activated sludge (RAS) from the secondary clarifiers. Screw pumps move the influent and RAS into an anoxic selector before the mixture enters the oxidation ditch. The oxidation ditch effluent from the primary ditch is sent through a pipe to the secondary clarifiers. Clarifier effluent is chlorinated prior to release to the Puget Sound. RAS is recycled to the anoxic selector. Sludge is wasted to another oxidation ditch, or wasted directly to a coarse air aerobic digester for bacterial reduction prior to solids handling activities. Anoxic conditions in the ditch are established by turning off the aeration each afternoon and then using ORP measurement to determine when the nitrate is depleted and restart aeration (usually 5 to 6 hours after aeration is turned off). General plant operating parameters are given in Table A5.2.

Table A5.2 Olympus Terrace Activated Sludge Design and Operating Parameters for August 1998

Parameter	***Value***
Aeration Detention Time, days	0.81
Type of Aeration	Surface aeration
Type of Anoxic Mixing	Mechanical Mixing
Typical MLSS, mg/L	2217
Typical MLVSS, mg/L	1884
VSS/TSS Ratio	0.85
Ditch Volume, MG	1.1
Anoxic Volume, MG	1.1
F/M ratio	0.16
SVI	137-150
Sludge Production, lb/day	1644.4
SRT, days	13

Aeration detention time is based on actual flow. The purpose of the ORP controlled aeration is to reduce energy costs and help control filaments. The plant uses one aerobic-anoxic cycle a day. The anoxic cycle usually lasts 4-6 hours a day. The anoxic cycle has reduced aeration costs by about 20%. The plant has observed less filamentous bulking characteristic and achieved lower SVI's since instituting the ORP controlled aeration. However, filamentous growth still occurs. The most prevalent filamentous population

now observed are the *Nocardi* and Type 0041 bacteria. These bacteria are generally associated with low DO levels and long SRTs. Sodium hypochloride (12% bleach) is used to further control filamentous populations at a dose of approximately 300 gal/month. Mixed liquor grab samples are analyzed daily for filamentous bacterial growth. When the filamentous populations start to establish themselves, sodium hypochloride is dripped unto the mixed liquor for 3 to 4 days to kill the filaments. The slow application of the sodium hypochloride has proven effective in killing the filaments without affecting the effluent quality during application.

Sludge wasting occurs daily from the RAS. The RAS TSS concentration is determined during each daily wasting cycle. The RAS concentration is relatively stable during the week but drops off during the weekend due to a large reduction in the industrial and commercial wastewater received. The plant's SRT is controlled to provide a certain oxygen utilization rate (OUR) in grab aeration tank mixed liquor samples. The target OUR for the plant is 10-15 mg O_2/L-hr. Generally, the plant operates an SRT of 12-15 days in the summer and 20-25 days in the winter.

The plant currently uses chlorine for disinfection. The district has approved a switch to UV disinfection by the year 2000.

A.2 Olympus Terrace Wastewater Characteristics

A summary of the plant's influent wastewater parameters for July and August 1998 are provided in Table A5.3.

TABLE A5.3 Olympus Terrace Average Wastewater Influent Characteristics

Parameter	July 1998 Ave	August 1998 Ave
Daily Flow, MGD	1.45	1.35
CBOD Concentration, mg/L	179	192
COD Concentration, mg/L	499	513
Influent pH	7.77	7.79
Influent DO, mg/L	5.7	5.3
Influent Temperature, °C	19.4	20.5
Influent TSS, mg/L	222	179
Influent VSS, mg/L	n.a.	n.a.

Olympus Terrace receives wastewater from a variety of different sources including domestic, commercial, and industrial sewage from Payne Airfield (Boeing Aircraft Assembly Plant), an electroplating plant, the county ferry system, and a fish packaging plant. The electroplating plant does have its own pretreatment facility, but has released slugs of metals (Cr, Cu, Zn) and acids from process upsets. The ferry system dumps very septic sewage with some seawater into the collection system at 2 to 3 a.m. creating some hydrogen sulfide problems.

Due to the large elevation changes between the WWTP and the short distance to its wastewater sources, much of the domestic wastewater has relatively short travel period within the collection system. This coupled with some ground water infiltration has resulted in relatively high levels of dissolved oxygen (DO) in the plant's influent wastewater.

A.3 Olympus Terrace WWTP Treatment Performance Data

Table A5.4 summarizes Olympus Terrace's treatment performance during July and August 1998:

Table A5.4 Olympus Terrace's Average Monthly Treatment Performance

during August and September 1998

Parameter	July 1998 Ave	August 1998 Ave
Effluent CBOD, mg/L	5.8	5.1
% CBOD Removal, mg/L	97	97
Effluent COD, mg/L	42.5	31
Effluent NH_3, mg/L	1.9	2.3
Effluent pH	7.11	7.04
Effluent DO, mg/L	7.5	7.5
Effluent TSS, mg/L	6.6	6.7

Olympus Terrace easily met its permit requirement during both months. This WWTP also meets its permit requirements throughout the year on a continuous basis. The plant also has substantially reduced its aeration requirements through ORP controlled aeration. This plant is not financially limited in treatment operations and runs its processes to try to get the best effluent conditions possible with existing facilities.

B. Snoqualmie Falls WWTP

The Snoqualmie Falls WWTP plant is located approximately 30 minutes East of Seattle in Snoqualmie Falls, Washington. A new facility was started in 1998 to replace a former lagoon treatment process. The current permit allows the plant to discharge treated wastes into the Snoqualmie River meeting the effluent limitations in Table A5.5.

TABLE A5.5 Snoqualmie Falls Effluent Permit Requirement

Parameter	*Weekly Average*	*Monthly Average*
Flow		0.23 MGD
CBOD	45 mg/L, 87 lbs/day 85% Influent BOD removal	30 mg/L, 58 lbs/day 85% Influent BOD removal
TSS	90 mg/L, 173 lbs/day 85% Influent BOD removal	60 mg/L, 115 lbs/day 85% Influent BOD removal
Fecal Coliforms	400/100 mL	200/100 mL
PH	$8.5 \geq$ pH ≥ 6.5	
Total Residual Chlorine	190 µg/L	65 µg/L 0.12 lb/d
Ammonia	15 mg/L	8.7 mg/L 16.7 lb/d
Copper	35 µg/L	20 µg/L 0.038 lb/d

Snoqualmie's current permit reflects the discharge requirements instituted when lagoon treatment was used. The existing permit expires in November 1999. A new permit is under review and will incorporate a higher average monthly flow to allow the plant to treat average monthly flows of 1 MGD predicted based on current and projected growth in the area.

B.1 Treatment Processes

The overall plant treatment processes are summarized in Figure A5.2.

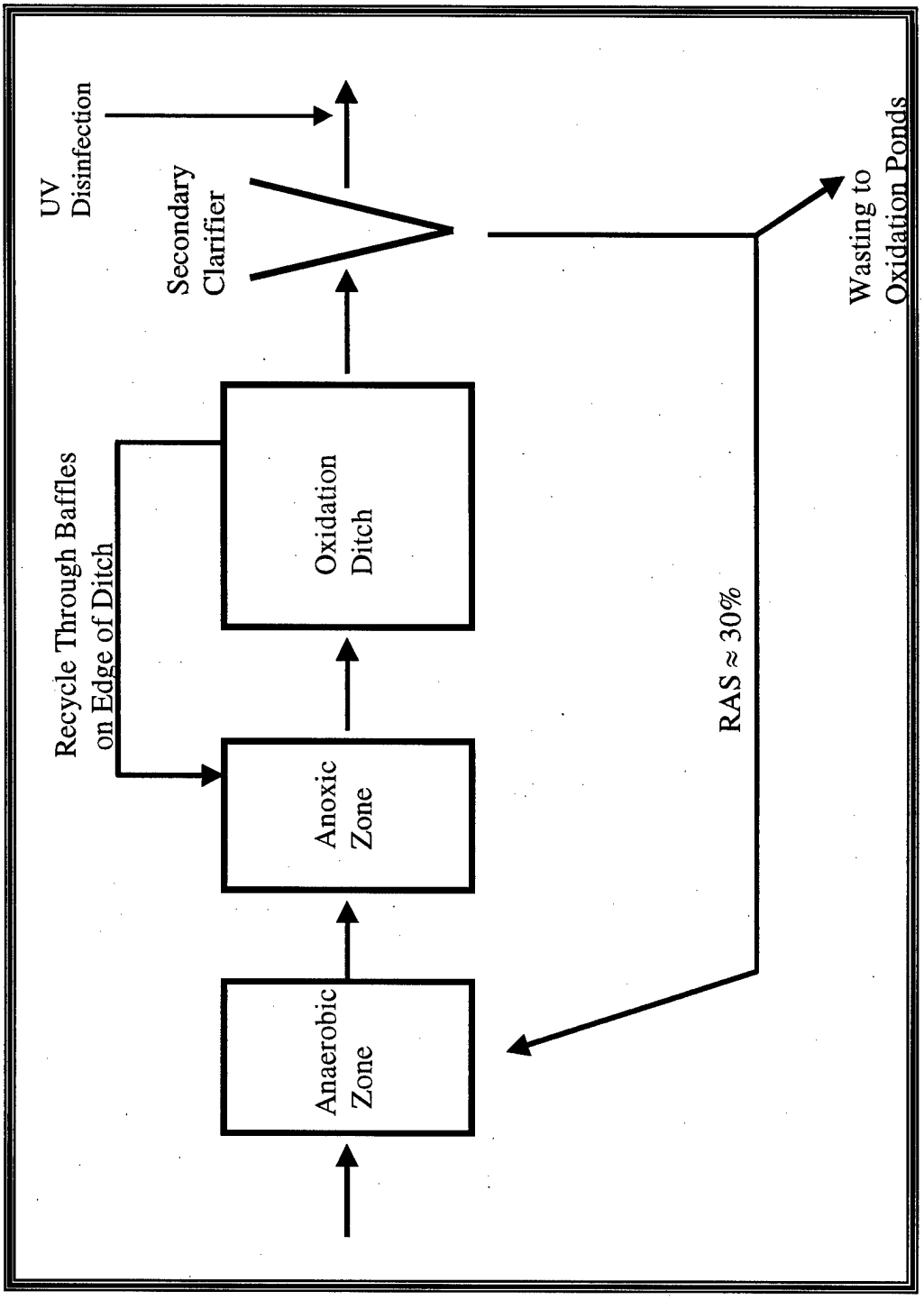


Figure 5A.2 Snoqualmie Falls WWTP Schematic

Snoqualmie Falls uses an anaerobic-anoxic-aerobic activated sludge process. No primary clarification is used. Wastewater is screened and degritted at the headworks. The wastewater is then mixed with the RAS and enters an anaerobic zone, then the mixed liquor moves to a single stage anoxic zone, and then into the oxidation ditch. Plant effluent is disinfected by UV treatment prior to discharge to the Snoqualmie River. All sludge wasting is to two holding ponds that are mixed and aerated with XX hp mixers. Currently, no offsite disposal of solids is required. The plant also has chlorination facilities intended for use with a water reuse program under development. General Operating parameters are given in Table A5.6 below:

Table A5.6 Snoqualmie Falls Activated Sludge Design and Operating Parameters for August 1998

Parameter	***Value***
Aeration Detention Time, days	3.6
Type of Aeration	Surface aeration
Type of Anoxic Mixing	Mechanical Mixing
Typical MLSS, mg/L	2217
VSS/TSS Ratio	n.a.
Ditch Volume, MG	1.04
Anoxic Volume, MG	0.26
F/M ratio	0.066
SVI	n.a.
Sludge Production, lb/d	284.7
SRT, days	23

Aeration detention time is based on actual flows observed. The F/M ratio is the design ratio. Sludge wasting occurs from the RAS line. Wasting is conducted on a daily basis. TSS quantification is conducted 2 to 3 times a week and the weekly average is used for calculating pounds wasted. The plant has not established an optimum or target mean cell residence time. The SRT has being varied between 60 and 10 days to determine it's effect on plant performance. System SRT was optimized through monitoring the plants effluent quality with SRTs ranging from 10 to 60 days. The best quality effluent was observed with an SRT of 60 days in terms of effluent BOD, TSS, nitrate, and ammonia removal rates. Based on this the plant changed the operating SRT to 60 days in

September 1998. SVI, MLVSS, and F/M parameters are not currently being analyzed and tracked at the plant. Sludge production, SRT, and other parameters used to determine active biomass are available but are not considered reliable since plant has yet to be truly operated under 'steady state' conditions in terms of plant parameters and complete analytical testing is not conducted at the plant.

B.2 Snoqualmie Falls Wastewater Characteristics

A summary of the plant's influent wastewater parameters for July and August 1998 are provided in Table A5.7 below.

TABLE A5.7 Snoqualmie Falls Average Wastewater Influent Characteristics

Parameter	July 1998 Ave	August 1998 Ave
Daily Flow, MGD	.277	.292
CBOD Concentration, mg/L	50.2	76.26
COD Concentration, mg/L	n.a.	n.a.
Influent pH	7.27	7.14
Influent DO, mg/L	2.51	2.64
Influent Temperature, °C	21.4	22.3
Influent TSS, mg/L	73.80	85.75
Influent VSS, mg/L	n.a.	n.a.

Snoqualmie Falls receives wastewater mainly from domestic sources in the area. The Weyerhauser company helped fund this plant for future development planned within the area. The golf course and housing areas under construction are expected to greatly increase domestic and commercial wastewater contributions within the next two years.

B.3 Snoqualmie Falls WWTP Treatment Performance Data

Table A5.8 summarizes the plant's treatment performance during the months of July and August 1998.

Table A5.8 Snoqualmie Falls Average Monthly Treatment Performance during August and September 1998

Parameter	July 1998 Ave	August 1998 Ave
Effluent CBOD, mg/L	1.4	1.5
% CBOD Removal, mg/L	97	98
Effluent COD, mg/L	n.a.	n.a.
Effluent NH_3, mg/L	0.157	0.152
Effluent pH	7.3	7.1
Effluent DO, mg/L	6.25	5.76
Effluent TSS, mg/L	2.0	1.0

This WWTP meets it's current permit requirements.

C. Chamber's Creek WWTP

The Chamber's Creek WWTP plant is located in Steilacoom, just South of Tacoma, Washington. The plant's NPDES permit allows the plant to discharge treated wastes into the Puget Sound meeting the effluent limitations in Table A5.9 below:

TABLE A5.9 Chamber's Creek Effluent Permit Requirements

Parameter	*Weekly Average*	*Monthly Average*
Flow		18 MGD
BOD	40 mg/L, 6005 lbs/day 85% Influent BOD removal	25 mg/L, 3750 lbs/day 85% Influent BOD removal
TSS	45 mg/L, 6760 lbs/day 85% Influent BOD removal	30 mg/L, 4500 lbs/day 85% Influent BOD removal
Fecal Coliforms	400/100 mL	200/100 mL
PH	$9.0 \geq$ pH ≥ 6.0	
Total Ammonia	Report Only	Report Only

C.1 General Plant Characteristics

The overall plant treatment processes are summarized in Figure A5.3.

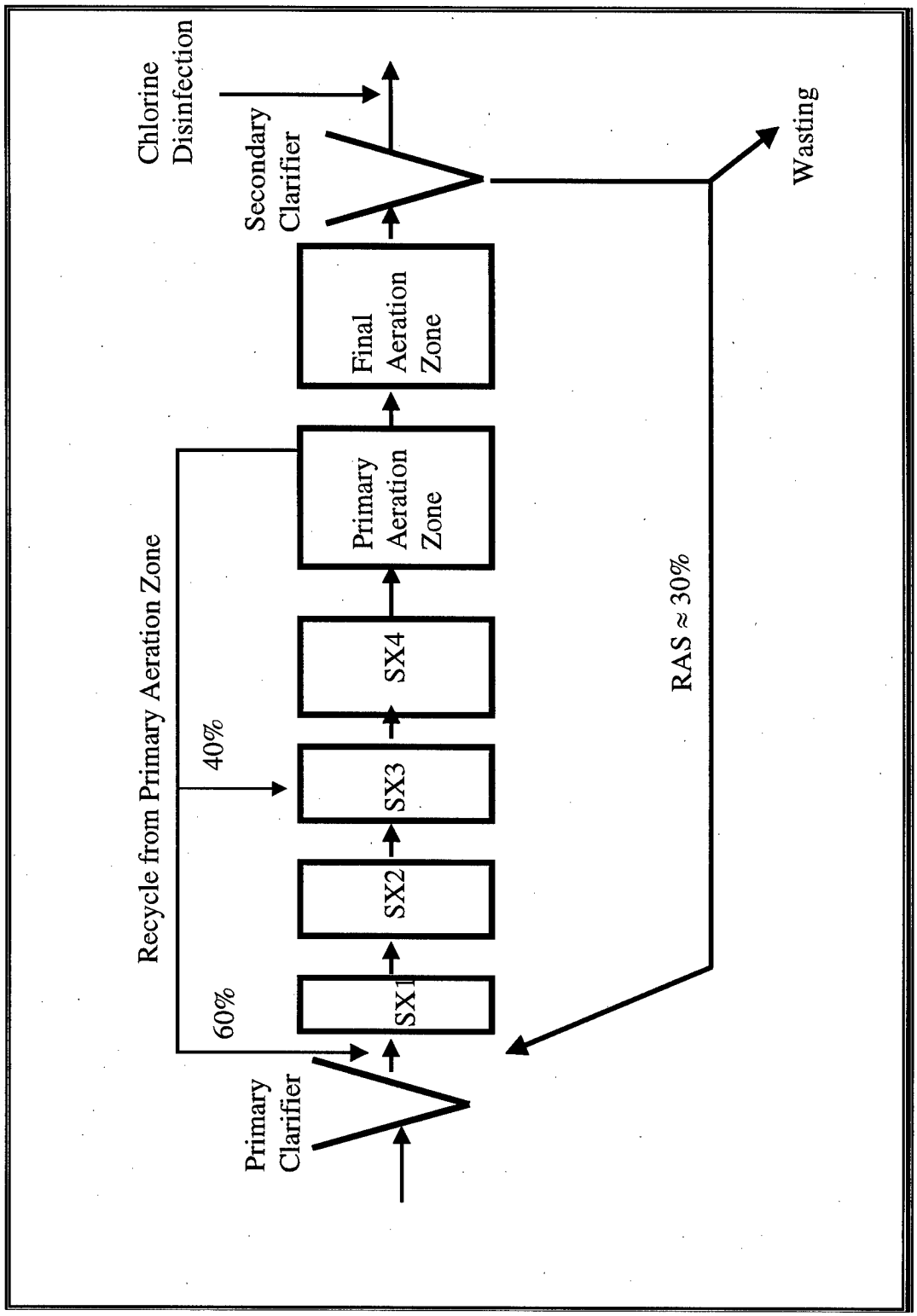


Figure 5A.3 Chamber's Creek WWTP Schematic

Chamber's Creek has primary treatment followed by an anoxic-aerobic activated sludge treatment process. The anoxic zone consists of four anoxic stages that are mixed by fine bubble diffusers at a rate that provides mixing but maintains DO concentrations below 0.1 mg/L The aeration zone consists of one large stages from which mixed liquor is recycled back to the first and third anoxic zones and a second smaller zone to remove remaining carbonaceous demand and purge the wastewater of nitrogen gas. Plant effluent is chlorine disinfected prior to discharge directly into Puget Sound. Waste sludge is treated by anaerobic digestion and then dewatered by belt presses before final disposal. General plant operating parameters are given in Table A5.10 below:

Table A5.10 Chamber's Creek Activated Sludge Design and Operating Parameters for August 1998

Parameter	***Value***
Aeration Detention Time, days	0.38
Type of Aeration	Fine air diffusion
Type of Anoxic Mixing	Fine air diffusion
Typical MLSS, mg/L	1587
Typical MLVSS, mg/L	1317
VSS/TSS Ratio	0.83
Aerobic Volume	3.38
Anoxic Volume	0.61
Stage 1 Detention Time, min	6.1
Stage 2 Detention Time, min	5.5
Stage 3 Detention Time, min	8.7
Stage 4Detention Time, min	15.1
F/M ratio	0.32
SVI	166
Sludge Production, lb/day	12614
SRT, days	4.4

Aeration detention time is based on actual flows. Approximately 50% of the primary aeration effluent is recycled back to the staged anoxic zones. The recycle is split 40-60 to the first anoxic stage and the third anoxic stage. RAS is mixed with the wastewater in the 1st Anoxic Zone. Sludge wasting occurs daily from the RAS line. Aeration air sparging rates are controlled by DO set points within the treatment train. DO probe set points for

the anoxic zones is 0.0 mg/l, 1.8 mg/L for the primary aeration zone, and 2.0 mg/l for the second aeration zone. Target SRT is 4.3 days during summer months.

C.2 Chamber's Creek Wastewater Characteristics

A summary of the plant's influent wastewater parameters for July and August 1998 are provided in Table A5.11 below:

TABLE A5.11 Chamber's Creek Average Wastewater Influent Characteristics

Parameter	July 1998 Ave	August 1998 Ave
Daily Flow, MGD	9.37	8.85
CBOD Concentration, mg/L	318	320
COD Concentration, mg/L	669	698
Influent pH	7	7
Influent DO, mg/L	1.2	1.7
Influent Temperature, °C	19.1	19.1
Influent TSS, mg/L	270	219
Influent VSS, mg/L	242	194

Chamber's Creek receives wastewater mostly from domestic and commercial sources. The plant receives wastewater from many distant sites and part of its wastewater has long collection system residence time.

C.3 Chamber's Creek WWTP Treatment Performance Data

Table A5.12 summarizes the plant's performance during the months of July and August 1998.

Table A5.12 Chamber's Creek Performance during August and September 1998

Parameter	July 1998 Ave	August 1998 Ave
Effluent CBOD, MGD	6.7	3.3
% CBOD Removal, mg/L	97	98
Effluent COD, mg/L	38	41
Effluent NH_3, mg/L	23.4	28.4
Effluent NO_3, mg/L	0.01	0.11
Effluent NO_2, mg/L	0.66	0.66
Effluent ALK, mg/L as $CaCO_3$	188	244
Effluent pH	7.3	7.5
Effluent DO, mg/L	6.2	5.4
Effluent TSS, mg/L	6.8	3.6

This WWTP consistently meets all daily, weekly, and monthly NPDES permit effluent requirements.

D. Lacey, Olympia, Tumwater, and Thurston County (LOTT) WWTP

The LOTT WWTP plant is located in Olympia, Washington. The plant provides wastewater collection and treatment to the cities of Lacey, Olympia, and Tumwater as well as the majority of Thurston County. The plant's NPDES permit allows the plant to discharge treated wastes into Budd Inlet (sensitive tract of the Puget Sound) meeting the effluent limitations in Table A5.13.

Table 4.13 LOTT Effluent Permit Requirements

Parameter	*Weekly Average*	*Monthly Average*
Flow		22 MGD
BOD	45 mg/L, 8256 lbs/day 85% Influent BOD removal	30 mg/L, 5504 lbs/day 85% Influent BOD removal
Seasonal BOD (Apr-Oct)	30 mg/L, 5504 lbs/day 85% Influent BOD removal	20 mg/L, 3670 lbs/day 85% Influent BOD removal
TSS	45 mg/L, 7898 lbs/day 85% Influent BOD removal	30 mg/L, 5265 lbs/day 85% Influent BOD removal
Fecal Coliforms	400/100 mL	200/100 mL
PH	$9.0 \geq$ pH ≥ 6.0	
TIN	3.0 mg/L	
Total Ammonia	31 mg/L	22 mg/L
Total Recoverable Copper	7.5 µg/L	6 µg/L

D.1 Treatment Processes

The overall plant treatment processes are summarized in Figure A5.4

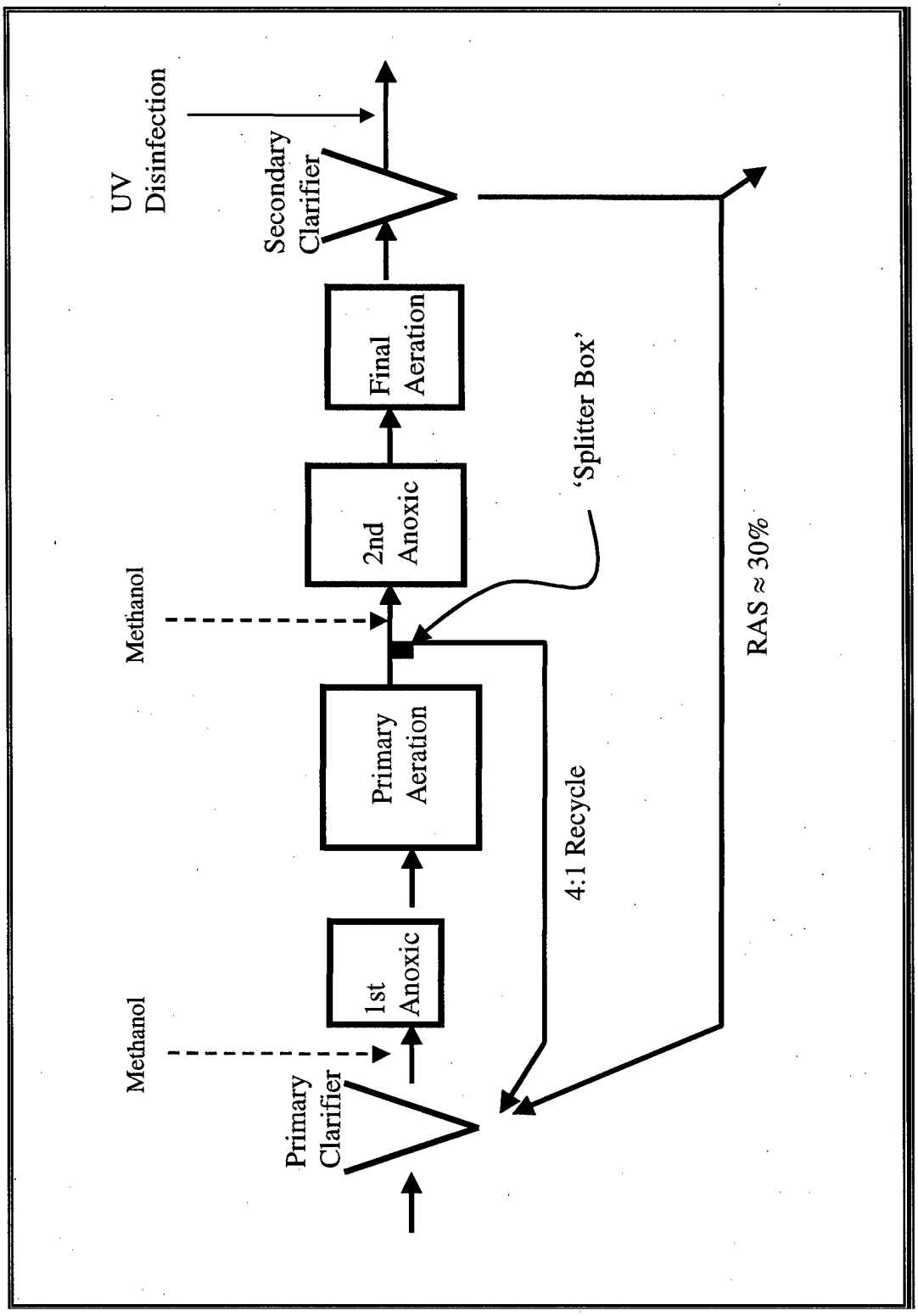


Figure 5A.4 LOTT WWTP Schematic

LOTT has primary treatment followed by a staged anoxic-aerobic–staged anoxic–aerobic activated sludge treatment process. Each anoxic zone consists of five stages that are mechanically mixed. All anoxic and aerobic treatment zones are enclosed. Plant effluent is UV disinfected prior to discharge into Budd Inlet. Waste sludge is treated by anaerobic digestion and then dewatered by centrifuges before final disposal. General plant operating parameters are given in Table A5.14 below:

Table A5.14 LOTT WWTP General Operating Parameters for August 1998

Parameter	*Value*
Aeration Detention Time, days	0.79
Type of Aeration	Fine air diffusion
Type of Anoxic Mixing	Mechanical Mixing
Typical MLSS, mg/L	2339
Typical MLVSS, mg/L	1871
VSS/TSS Ratio	0.80
1st Aerobic Zone Volume, MG	6.45
2nd Aerobic Zone Volume, MG	0.54
1st Anoxic Zone Volume, MG	1.74
2nd Anoxic Zone Volume, MG	1.62
F/M ratio, g COD/gTSS-d	0.34
SVI	85
Sludge Production, lb/d	10495
SRT, days	11.6

Aeration detention time is based on actual flows. Primary aeration effluent is recycled at a ratio of 4:1 back to the 1st Anoxic Zone. Sludge wasting occurs from the RAS line. The RAS is mixed with the wastewater in the 1st anoxic stage of the first anoxic zone of treatment. Methanol periodically used as an RBCOD additive to enhance nitrogen removal to met permit requirements. Methanol addition is usually required for only weeks out of each year. The plant's operators use nitrate concentration at the 'splitter box' as a trigger for methanol addition. If the nitrate concentration is less than 4.5 mg/L at the 'splitter box' then no methanol is added to any treatment processes. If nitrate concentration at the 'splitter box' is greater than 4.5 mg/L, but less than 6.5 mg/L then methanol is added to the 2nd Anoxic Zone. If nitrate concentration is greater than 6.5 mg/L at the 'splitter box' then methanol is added to both anoxic zones. Soda ash is stored

on site and added when necessary to control pH conditions. Thus far, the plant has not had to add soda ash with the methanol to due to high wastewater alkalinity levels. When methanol is added to the anoxic zone it is sprayed unto the mixed liquor in the first stage of each zone.

The operators reported that they have conducted tests to see how quickly the methanol is utilized in the anoxic tanks. From their testing they found that methanol is removed from the mixed liquor almost immediately within the first anoxic stage. Larry Ekstrum, the plant's lead operator stated that the plant's ability to meet its seasonal TIN permit requirements has been directly linked to local brewery's wastewater output. When the brewery is operating, the WWTP often cannot meet permit requirements without methanol addition. When the brewery and soda factory are not active, the plant must add methanol to meet permit requirements. The WWTP has remote monitors for the brewery discharges to improve plant operations and response times to changing wastewater characteristics.

D.2 LOTT Wastewater Characteristics

A summary of the plant's influent wastewater parameters for July and August 1998 are provided in Table A5.14 below:

Table A5.15 LOTT Average Wastewater Influent Characteristics

Parameter	July 1998 Ave	August 1998 Ave
Daily Flow, MGD	9.37	8.85
CBOD Concentration, mg/L	318	320
COD Concentration, mg/L	669	698
Influent pH	7.0	7.0
Influent DO, mg/L	1.2	1.7
Influent Temperature, °C	19.1	19.1
Influent TSS, mg/L	255	269
Influent VSS, mg/L	211	236

LOTT also receives wastewater from a variety of sources. These include domestic, industrial, soda factory, and brewery wastes. Brewery waste flows and compositions are considered by the operators of the plant to be the most important aspect of the influent wastewater.

D.3 LOTT WWTP Treatment Performance Data

Table A5.16 summarizes the plant's performance during the months of July and August 1998.

Table A5.16 LOTT WWTP's Average Monthly Treatment Performance during August and September 1998

Parameter	July 1998 Ave	August 1998 Ave
Effluent CBOD, mg/L	2.86	2.72
% CBOD Removal, mg/L	98.4	98.9
Effluent COD, mg/L	33	25
Effluent NH_3, mg/L	0.126	0.093
Effluent NO_3, mg/L	1.61	1.52
Effluent NO_2, mg/L	0.012	0.093
Effluent PO_4, mg/L	2.84	5.04
Effluent pH	7.2	7.1
Effluent DO, mg/L	5.9	5.7
Effluent Alkalinity, mg/L as $CaCO_3$	149	139
Effluent TSS, mg/L	6.43	6.09

This WWTP met its stringent permit requirements. This WWTP also produces the highest quality effluent in terms of nutrient loading of the four WWTPs.